Are moorland invertebrates resilient to fire?

Plate 1 (previous page): Tabanidae on buttongrass flower-head

Are moorland invertebrates resilient to fire?

by

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Statement of co-authorship

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Shifts in composition of monthly invertebrate assemblages in moorland differed between lowland and montane locations but not fire-ages

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Abstract

Fire is an integral part of ecosystem management in many biomes around the world. This is particularly the case for the fire-prone and highly flammable moorlands of the Tasmanian Wilderness World Heritage Area. To maintain ecosystem processes and to conserve biodiversity, fire management in protected areas needs to be based on an understanding of the consequences of fire regimes. Invertebrates are a very diverse group of organisms that are critical for sustaining ecosystem processes. However, world-wide, there are few well-designed, long-term studies involving a broad range of taxa that have investigated the resilience of invertebrates to fire, and none in Tasmanian moorlands.

I investigated the resilience of ground- and foliage-active invertebrate assemblages to low-intensity fire in low and moderate productivity moorlands, and identified predictors of assemblage change and its return to the pre-fire state. I used two complementary research designs: (1) a before-after-control impact design that aimed to minimise spatial variation, and (2) a replicated space-for-time design that provided insight into longer-term invertebrate responses to fire, and had a broad spatial scale. Species-level resolution for a broad range of invertebrate groups was used.

I found that ground- and foliage-active invertebrate assemblages in moorland were resilient to single fires and to a fire regime with a mean fire interval of at least 24 years in low productivity moorlands and 37 years in moderate productivity moorlands. Low-intensity fire in moorland altered the composition of invertebrate assemblages because many taxa had low abundance in early successional stages. However, contrary to expectations, I found that some ground-active taxa had higher abundance in early successional stages. Very few taxa were absent from early (<3 years) successional stages and none were absent from later successional stages. Return to the pre-fire state was highly deterministic and mediated by the return of vegetation and other habitat elements to the pre-fire state. Vegetation density and several other environmental variables were better predictors of assemblage response to fire than time-since-fire. Invertebrate assemblages in low productivity

moorlands generally took twice as long to return to the pre-fire state than assemblages on moderate productivity soils.

Trophic status, position in the environment, association with decaying vegetation and development type were species traits that best predicted invertebrate composition response to fire, but only explained 19% of the variation. These species traits require further investigation before they can be considered useful and other traits need to be considered. Higher level taxa were found to be effective surrogates for species-level identification in representing patterns in invertebrate assemblage structure and detecting the effects of fire.

To improve fire management for fauna, further research on functional traits and the influence of spatial and temporal arrangement of fire regimes on biota is required.

Plate 2 Cup ringtail damselfly *Austrolestes psyche* (Hagen in Selys) (Odonata: Lestidae)

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Plate 3 Burrowing crayfish *Ombrasticoides huonensis* Hansen and Richardson (Decapoda: Parastacidae), found deceased post-fire

Table of Contents

List of Plates

Plate 4 Tassie hopper *Russalpia longifurca* Key (Orthoptera: Acrididae)

Plate 5 Wingless grasshopper *Phaulacridium vittatum* (Orthoptera: Acrididae)

Chapter 1

General Introduction

1.1 Introduction

Fire is an important ecological process shaping landscapes and their biota in many terrestrial ecosystems around the world and is widely used for ecosystem management (Gill et al. 2002; Moretti et al. 2002; Parr and Chown 2003). While some ecosystems are particularly sensitive to the impacts of fire, such as coniferous forests (Holz et al. 2015) and European silver fir forests (Tinner et al. 2005), it is increasingly apparent that fire suppression or exclusion in many ecosystems causes significant changes in vegetation structure and composition and can result in the local loss of some plant and animal species. The associated increase in fuel loads can also create unacceptable risks to people, infrastructure and adjacent fire sensitive ecosystems. For example, fire suppression in ponderosa pine forests of southwest United States of America following Euro-American settlement changed the community from park-like landscapes to dense pine regeneration with decreases in growth and diversity of both herbaceous and woody plants (Covington *et al.* 1997), and fire suppression in mixed conifer stands in the Pacific northwest United States of America, initiated in the early 1990s to prevent loss of wood resources, resulted in greater fuel loads and increased the chances of high intensity catastrophic wildfire (Niwa and Peck 2002). In northern savannas of Australia, the removal of anthropogenic fire following European settlement resulted in structural changes in the vegetation, particularly an increase in the biomass of woody plant species (Dyer et al. 1997; Hoffmann 2003) and similar changes within savannah landscapes with a reduction in fire frequency have been observed around the world (Bullock et al. 1998). In moorlands of western Tasmania, a reduction in fire-frequency following European settlement led to periods of vegetation growth and fuel build-up followed by regional-scale fires that spread into and caused the loss of significant areas of fire-sensitive vegetation (Marsden-Smedley and Kirkpatrick 2000). Where the exclusion of fire from ecosystems has caused unintended

consequences there have been calls to re-introduce fires for the benefit of ecosystem management, to better manage biodiversity and to reduce the chances of catastrophic fires. The challenge for land managers is to determine appropriate fire regimes within a context of competing objectives.

Better knowledge of the responses of ecosystem components, such as plants and animals, to the different components of a fire regime, such as fire interval, fire season and fire intensity, is required for fire management planning. However, because of the diversity of species and their life histories, and the interactions that can occur between their responses to fire regime components, it is hardly surprising that for most ecosystems such information is far from complete. Grouping plants and animals on the basis of their responses to fire has been recommended as one of the ways forward for fire management planning for biodiversity conservation (Keith 1996; Gill et al. 2002; Driscoll et al. 2010). To reduce the complexity of biodiversity conservation, Keith et al. (2002b) suggested that "A good strategy to conserve *all* populations would be based on detailed knowledge, management action and monitoring of a *few* species, particularly those species or groups of species which have traits that render them most susceptible to decline across any of the possible fire regimes". Functional trait classifications have been developed for plants in fire-prone regions based on methods of persistence or re-establishment after fire (Noble and Slatyer 1980; Keith and Bradstock 1994). While there have been some attempts at functional classifications for animals (Andersen 1995; Friend 1995a; York 1999b; Keith et al. 2002a), they are poorly developed and have not been applied more broadly (Gill et al. 2002; Keith et al. 2002b; Moretti and Legg 2009; Driscoll et al. 2010). Unlike plants in which life history characteristics exert critical influence over population dynamics, the principal functional attributes for animals relate to dispersal, behaviour and resource use (Keith et al. 2002b). However, as Whelan *et al.* (2002) conclude, fire responses will be site-specific, making it almost impossible to predict the effects of a particular fire (or sequence of fires) on any species. They suggested that instead of seeking general fire response patterns the focus should be on using field experiments to understand how life-cycle processes produce various patterns of response (mortality in fire, recolonisation, survival and establishment of individuals after fire, reproduction and population growth) and

how the factors that mediate them (e.g., habitat quality, nest sites, nutrient and food availability and predation) respond to different fire, landscape and climatic characteristics. Keith *et al.* (2002a) attempted to apply the system presented by Whelan *et al.* (2002) to a small selection of heathland animals. While they found that it may be feasible to define attributes of heathland animals in such a way as to predict fire responses, considerable autecological data are needed to apply such a classification and, even for well-known species, predictions about fire response may be sensitive to uncertainty in a single attribute. These fire response attributes also depend partly on extrinsic factors that are not characteristics of the organism, but characteristics of habitat and fire. Keith et al. (2002a) concluded that the effects of fire frequency on animal populations, particularly invertebrates, requires further research, particularly the development of novel life-history approaches.

Invertebrates are a tremendously diverse group of organisms and play an essential role in the maintenance of ecological processes (Kim 1993; Woodward 1994; McGavin 2000). In addition to their important role in ecosystems, the high densities and short generation times of many taxa make them ideal subjects for replicated fire studies (Whelan et al. 2002). Despite this, surprisingly little is known about the basic responses of invertebrates to fire in many ecosystems (Gill et al. 2002; Whelan et al. 2002; Parr and Chown 2003; New 2014). Where research has been undertaken, the responses exhibited by invertebrates tend to be variable, often difficult to detect and with few consistent patterns (Tscharntke and Greiler 1995; Friend and Williams 1996; Whelan et al. 2002; Parr and Chown 2003; New 2014). These variable responses could result from several issues associated with fire studies on invertebrates (Friend 1995a; Swengel 2001; Whelan et al. 2002; Munro et al. 2009; New 2014):

1. Invertebrates present significant challenges for species-level identification especially if a broad range of taxa are to be investigated. Previous fire studies have typically investigated a particular order or family of invertebrates or have identified a broad range of taxa to order level only. Given that invertebrates exhibit a wide range of life histories and occupy many different

types of habitats even within a particular order or family, it is understandable that few consistent response patterns to fire have emerged.

2. A wide variety of sampling methods are used to estimate invertebrate diversity, abundance and activity such as pitfall trapping, sweep netting, soil cores and litter samples. Different sampling methods can result in different observed invertebrate responses to fire because each method has its own sampling bias or can be deployed differently.

3. Invertebrate populations change both temporally and spatially. Many previous studies investigating invertebrate responses to fire lack rigor because of inadequate controls and replication (both temporally and spatially) and with little or no information on pre-fire patterns in diversity and abundance. Most such studies have been short-term (<12 months), focussing on the immediate post-fire effects and did not track populations in relation to vegetation succession.

4. Key components of the fire regime such as fire intensity, season of burn and fire interval potentially influence the pattern of fire responses by invertebrates and add another layer of complexity.

It is therefore essential that experiments investigating the responses of invertebrates to fire are carefully designed, incorporating pre- and post-fire sampling of control and burnt sites.

Some consistent fire response patterns have emerged from invertebrate studies. Invertebrate assemblage composition changes following fire with most taxa declining markedly in abundance immediately after the event (see reviews by Warren et al. 1987; Friend 1995b; Whelan 1995; Swengel 2001; Whelan et al. 2002; New 2014). This is because fire often consumes much of the vegetation that invertebrates are dependent upon for food, feeding niches and shelter (Lawton 1983; Evans 1984). Some taxa increase after fire taking advantage of post-fire resources such as new oviposition sites and food. Pyrophilous taxa include

Coleoptera that are attracted to flames and smoke during or very soon after fire (New 2014) and some Formicidae and Orthoptera that take advantage of the early post-fire successional stage (e.g. O'Dowd and Gill 1984a; Bess et al. 2002; Hochkirch and Adorf 2007). Few studies are designed to determine the resilience of invertebrate assemblages to fire, that is, how long does species composition take to return to pre-fire levels, if at all. Those studies that have been thus designed, have shown that when vegetation and other environmental elements return to pre-fire conditions, the invertebrate assemblage composition returns to pre-fire levels of diversity, abundance and composition, particularly in fire-prone environments (Friend and Williams 1996; Andersen and Müller 2000; Bess et al. 2002; Moretti et al. 2004; Pyrke and Samways 2012b).

Many invertebrate studies investigating the effects of fire have used higher taxonomic groups instead of species-level identifications because of limited resources and/or lack of taxonomic expertise. However, conclusions drawn from such studies should be treated with caution because impacts on invertebrate richness and composition or individual species reactions to fire may be masked. One solution is to test the efficacy of higher taxa (i.e. groups of related taxa) as surrogates for species-level identification. In a review of almost 300 case studies, covering a range of environments and organisms, Bevilacqua et al. (2012) found that higher taxa up to at least family level can be successful in identifying patterns of community change due to natural or human disturbance. The performance of higher taxa is strongly influenced by the strength of the perturbation and the ratio of the number of higher taxa to the number of species and its variance (Olsgard et al. 1997; Bevilacqua et al. 2012; Neeson et al. 2013; van Rijn et al. 2015). Higher taxa surrogates perform best in communities in which a few common species are most abundant and in which the ratio of higher taxa to lower taxa is high (>0. 4) (Bevilacqua et al. 2012; Neeson et al. 2013). Ultimately the successful application of higher taxonomic surrogates will depend on the objectives of the study and a trade-off between time saved and value of the survey.

1.2 Buttongrass moorland

In Tasmania, there have been calls to increase the amount of planned burning in buttongrass moorlands for the benefit of local biodiversity and to reduce the risk of catastrophic fires burning across the landscape into fire sensitive communities and threatening human lives and property (Marsden-Smedley and Kirkpatrick 2000; Marsden-Smedley 2009). Buttongrass moorland is a treeless sedgeland in which the tussock sedge, commonly known as buttongrass, *Gymnoschoenus sphaerocephalus* (Brown) Hooker (Cyperaceae), is often dominant (Jarman et al. 1988) (Plate 6). It is a significant landscape feature of western Tasmania occupying over 500 000 ha and has been recognised as having world heritage value (Balmer et al. 2004). Buttongrass moorland is also a very flammable vegetation type (Marsden-Smedley 1998) and was burnt by hunter-gathers over thousands of years to facilitate easy passage and to encourage game (Thomas 1993). The current extent of buttongrass moorland appears to represent an anthropogenic disclimax extended far beyond its natural edaphic limits by fire (Jackson 1968). The 'ecological drift' model proposes that, in the absence of fire, buttongrass will become scrub and, eventually, rainforest (Jackson 1968). Since the cessation of Indigenous burning practices in western Tasmania following European settlement, the fire regime has been one with long periods without fire, resulting in vegetation growth and fuel build up followed by large conflagrations. This fire regime has been postulated to have caused the loss of significant areas of fire sensitive vegetation (Marsden-Smedley 1998; Marsden-Smedley and Kirkpatrick 2000). Marsden-Smedley and Kirkpatrick (2000) make a case that the maintenance of the ecological values of south-west Tasmania is likely to be best served by an integrated program of broad-scale ecosystem-management burning on about a 20 year rotation in association with tactical hazard-reduction burning on a 5–8 year rotation and fire suppression where appropriate. They argue that this fire regime (using planned low-intensity burns) would resemble putative Indigenous fire regimes. Such a program would significantly increase the area and frequency of planned burning in buttongrass moorlands compared to recent decades. Although one of the primary goals of applying this fire regime in buttongrass moorlands is the maintenance of plant and animal diversity, very little is known about the response of animals to fire in this

vegetation community (Driessen 2010). Indeed there has been very little systematic survey of the fauna values of buttongrass moorlands at all (Driessen 2006, 2008). Most fire research on fauna to date has been limited in scope, unpublished and or not well designed in terms of controls and replication. Arkel (1995) and Driessen (1999) investigated the effect of fire on small mammals, with the former comparing their diversity and abundance in buttongrass with different fire ages and the latter monitoring populations before and after a planned burn. Bryant (1991) surveyed the distribution and abundance of ground parrots, *Pezoporus wallicus* Kerr, 1792 (Psittaculidae), and related this to the fire age of buttongrass moorland. Chaudhry (2010) used a replicated space-for-time design to investigate the response of avifauna to post-fire succession. Greenslade and Driessen (1999) compared the diversity and abundance of invertebrates in buttongrass moorlands with different fire ages, and later used a paired design to compare Collembola communities between young and old regrowth moorlands (Driessen and Greenslade 2004). Green (2008, 2009) investigated the impact of fire on soil mites using a replicated space-for-time design. Nothing is known of the fire responses of most of the invertebrate fauna. Thus, there is a clear need for further research to understand the response of animal communities to fire in buttongrass moorlands to assist with management of this world heritage ecosystem. An investigation into invertebrate community succession was considered to be particularly useful because of their diversity, their importance for ecosystem function and because they have been poorly studied.

1.3 Thesis objective, questions and structure

The overall objective of the present study is to investigate the response and resilience of invertebrate assemblages in buttongrass moorlands following planned low-intensity burns and to identify predictors of assemblage change and its return to a pre-fire state. Specifically, the following questions were addressed:

1. Are moorland invertebrates resilient to fire?
2. Do moorland invertebrate assemblages change following fire and is there a pattern of succession?

3. Are there moorland taxa that are sensitive to fire?
4. Are functional traits important predictors of invertebrate response to fire?
5. Are higher taxa effective surrogates for species-level identification in representing patterns of assemblage change due to fire?

Chapters 2–6 have been written as stand-alone papers for publication. As a consequence there will be some repetition of information between chapters. Chapter 2 has been published in *Environmental Entomology* (Driessen et al. 2013). To minimise the number of printed pages, references cited in all chapters are collated at the end of this thesis.

Chapter 2 provides an understanding of the diversity of invertebrates in buttongrass moorlands and their seasonal variation in abundance and composition. This information provides guidance on the optimum time to conduct invertebrate surveys in buttongrass moorland where year-round surveys are not possible, and identifies the limitations associated with surveying at particular times of the year. Incorporated into the design of Chapter 2 is a comparison of invertebrate assemblages between young and old regrowth moorland throughout the year. This is one of very few studies that have investigated variation in seasonal activity across a large number of orders that have been identified to species or morphospecies level (Appendix 1; Plates 1–5, 11–12 and 18–23).

Chapters 3 and 5 address the overall objective of the thesis and questions 1–3. Chapter 3 uses a replicated space-for-time design at two locations over a broad spatial scale (19–25 sites over 150–900 km^2) that provides insight into long-term (1–65 years) invertebrate responses to fire. An unusual feature of the approach taken in Chapter 3 is that virtually all invertebrates are identified to species or morphospecies level. Chapter 5 uses the robust before-after-control-impact design within a small spatial scale (a single site within two locations) to monitor the response of invertebrates to fire. Although a broad range of invertebrate groups were targeted, limited time and funds meant that the analysis was conducted using ordinal-level identification. The efficacy of using higher taxa as surrogates for species-level identification (Question 5) is tested in Chapter 4. The study also tested several potential predictors of the effectiveness of higher taxonomic surrogates.

Chapter 6 uses a broad-range of invertebrate species across 21 orders to investigate whether functional traits are important predictors of invertebrate response to fire (Question 4).

In Chapter 7, I synthesise the results of the previous chapters in relation to questions presented above and discuss the implications of my research for future fire management and research.

1.4 Overview of study locations and sites

The Tasmanian Wilderness World Heritage Area in western Tasmania is a rugged, glacially modified mountainous region covered predominantly by moorland, rainforest, alpine heathland, grassland and eucalypt forest. Vehicular access is largely restricted to the margins of the world heritage area and only two major roads enter it. This limited access placed restrictions on study location selection. The present study was conducted at two locations; one in montane buttongrass moorlands in the vicinity of Lake St Clair on the Central Plateau and the other in lowland moorlands in the vicinity of Lake Pedder (Fig. 1.1). Moorlands at each location contain one of the two buttongrass moorland vegetation groups that are widespread in central and western Tasmania: blanket moor and eastern moor (Jarman et al. 1988). Blanket moor is the dominant moorland near Lake Pedder (Plate 7) and accounts for much of buttongrass moorland in western Tasmania where they are mostly underlain by Precambrian sediments on which form shallow infertile soils. Eastern moor is the dominant moorland near Lake St Clair (Plate 8) and is widespread in highland regions of central and western Tasmania where it is usually associated with dolerite geology with relatively fertile soils.

Moorlands in these locations were burnt by hunter-gatherers over thousands of years, putatively on a frequent, low-intensity basis (Marsden-Smedley 1998). Following the removal of indigenous people by Europeans in the nineteenth century, the moorlands have been subjected to bushfires caused by lightning strikes, pioneers, explorers, arsonists and land managers. Since the 1970s, low-intensity planned burns have been conducted by the Tasmanian Parks and Wildlife Service to manage the threat of bushfires.

The various components of the present study were undertaken at the sites shown for each location in Fig. 1.1.

Plate 6 Buttongrass tussock *Gymnoschoenus sphaerocephalus* (Brown) Hooker (Cyperaceae) in montane moorland

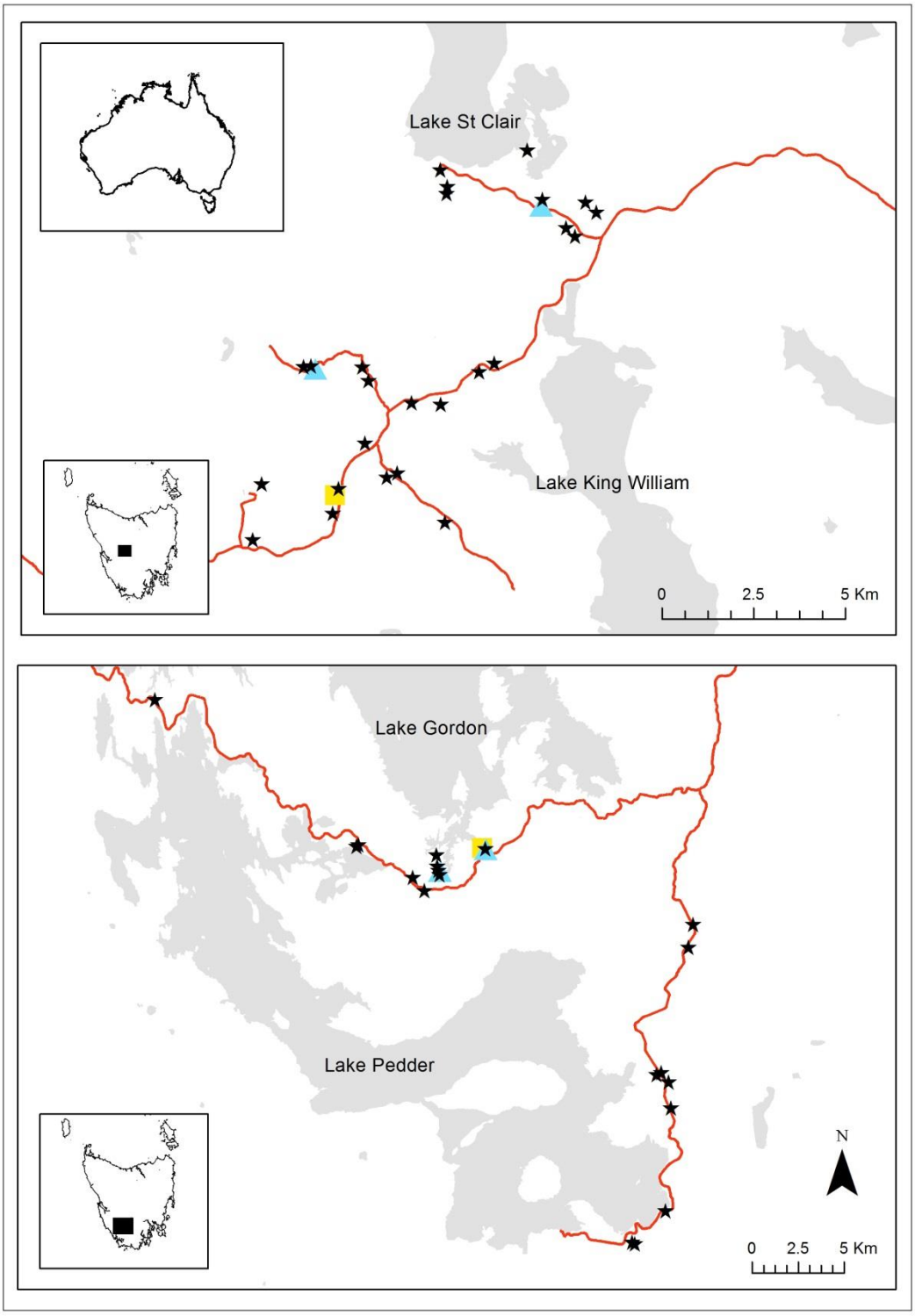

Fig. 1.1 Location of sites in lowland moorland (bottom) and montane moorland (top) used in the present study. Blue triangles: shifts in monthly invertebrate composition (Chapter 2). Black stars: invertebrate responses to fire: space-for-time study (Chapter 3). Yellow squares: invertebrate responses to fire: before-after-control study (Chapter 5). Shading = waterbodies. Solid red lines = roads.

Plate 7 Blanket moor near Lake Pedder (lowland moorland, low productivity)

Plate 8 Eastern moor near Lake St Clair (montane moorland, moderate productivity)

Chapter 2

Shifts in composition of monthly invertebrate assemblages in moorland differed between lowland and montane locations but not fire-ages.

2.1 Abstract

Understanding seasonal changes in invertebrate populations is important for understanding ecosystem processes and for conservation of invertebrate communities. Few studies have investigated variation in seasonal responses of multiorder and multispecies invertebrate assemblages. To determine whether temporal patterns in invertebrate assemblages and taxa vary between locations and vegetation age since burning, patterns of invertebrate occurrence were investigated monthly for 12 months in cool temperate buttongrass moorlands at two locations (lowland and montane) containing paired plots with different fire history (young and old regrowth). For both locations and fire-ages invertebrate taxon richness and abundance were generally higher during the warmer months than during the winter months. At the lowland location ground-active invertebrates were caught in greater numbers during winter than during summer owing to large numbers of Collembola. Each season had a distinct invertebrate assemblage. The invertebrate assemblages did not differ between young and old regrowth. The shifts in composition of monthly invertebrate assemblages between winter and summer differed between locations with assemblages in cooler months more dissimilar from warmer months at the montane location than the lowland location. Most taxa common to both locations had similar patterns of monthly occurrence but some taxa showed markedly different patterns. Mid- to late summer is the optimum time to conduct short-term surveys in buttongrass moorland to maximise species richness and abundance but short-term studies will miss significant components of the invertebrate community.

Key words: seasonality, elevation, invertebrate surveys, buttongrass moorland, Tasmania.

2.2 Introduction

Understanding temporal and spatial changes in populations of invertebrate assemblages is an important first step in gaining knowledge of ecosystem processes (Wolda 1979; Weeks and Holtzer 2000; Grimbacher and Stork 2009) and is a key component of the processes of documenting and maintaining biological diversity (Kim 1993; Barrow and Parr 2008). Knowledge of seasonal patterns of invertebrate occurrence in an ecosystem can also be valuable for planning the timing of experimental research, biodiversity surveys and taxon-specific surveys, particularly where time and resources are restricted. For examples, late-spring to early-summer was recommended as the optimal time to conduct short-term sampling programs for spiders in Mediterranean ecosystems as this was when species richness was greatest (Cardoso et al. 2007). In arid zone Australia, spring and autumn have been suggested as the most effective time to survey invertebrates because of high species richness and abundance (Palmer 2010).

Although seasonality in invertebrate abundance and activity is a well-known phenomenon particularly in temperate ecosystems (Denlinger 1980; Masaki 1983; Wolda 1988; Cardoso et al. 2007), quantitative descriptions of multi-species assemblages are remarkably few even in temperate ecosystems. This is because such studies require regular monitoring for at least a year and the identification and enumeration of many hundreds of species, often represented by many thousands of individuals. Consequently, the majority of studies that have been undertaken have documented invertebrate community seasonality either at the ordinal and or functional group level (e.g. Willis 1976; Lowman 1982; Woinarski and Cullen 1984; Bell 1985; Recher et al. 1996; Pinheiro et al. 2002; Anu et al. 2009) or the species and or morphospecies level within one or two orders (e.g. New 1979; Wolda and Chandler 1996; Gutiérrez and Menénendez 1998; Novotny and Basset 1998; Cardoso et al. 2007; Grimbacher and Stork 2009; Kishimoto-Yamada et al. 2010). Very few terrestrial invertebrate seasonality studies have attempted to identify a range of species within a broad sweep of orders or to investigate changes at the community level (e.g. Nielsen 1974; Abbott et al. 1984; Southwood et al. 2004; Palmer 2010) and even fewer have attempted to do this by comparing seasonality between different locations.

In temperate regions, some general patterns emerge from studies of seasonality of invertebrate activity and abundance, despite differences in taxa, survey methods, habitats, and level of identification. In nearly all studies covering a broad range of invertebrate groups, peak activity, abundance, biomass and or diversity occur during the warmer months, primarily during spring and summer (e.g. Nielsen 1974; New 1979; Lowman 1982; Baribeau and Maire 1983; Ohmart et al. 1983; Recher et al. 1983; Bell 1985; Masteller 1993; Gutiérrez and Menénendez 1998; Southwood et al. 2004; Cardoso et al. 2007) but also during autumn (e.g. Nielsen 1974; New 1979; Abraham 1983; Woinarski and Cullen 1984; Recher et al. 1996; Chatzaki et al. 2009). However, exceptions to this pattern do occur. For example, no clear seasonal pattern was found in the activity of flying insects in Australian heathlands (Pyke 1983). In the canopy of *Eucalyptus* forests of Western Australia, where summers are hot and dry, the abundance of several insect orders was found to be highest in winter and lowest in summer (Recher et al. 1996).

Within this general pattern of peak invertebrate activity during the warmer months, many studies have shown there is considerable variation between species within families and between families within orders in the timing of peak activity. Furthermore, this can vary from year to year and some taxa may have more than one peak of activity. For example, on various *Acacia* host plants in temperate Australia, different species of beetles were recorded peaking at different times during the warmer months with some species peaking in both spring and autumn (New 1979). Seasonal responses in invertebrate activity can occur on a range of scales with local scale influences (e.g., timing of flowering and leaf flush) on life-cycle events occurring within the context of intermediate scale (e.g. topographic complexity and elevation) and large scale (e.g. latitude) influences (Danks 2006, 2007). Whilst there have been previous studies comparing seasonality in terrestrial invertebrate assemblages at different scales, they are few in number. Most compare seasonality on large scales such as between different climatic zones (e.g. Lowman 1982; Baribeau and Maire 1983; Masteller 1993; Recher et al. 1996). Studies comparing seasonality in multispecies assemblages between local environments are rare and are taxonomically highly focussed. In northern Spain the flight period of butterfly assemblages was shorter and occurred later at higher

elevations than at lower elevations (Gutiérrez and Menénendez 1998). In a steppe ecosystem, seasonal changes in spider assemblages were found to differ between two habitat types (Weeks and Holtzer 2000).

Buttongrass moorland is a treeless sedgeland in which the hummock sedge, commonly known as buttongrass, *Gymnoschoenus sphaerocephalus* (Brown) Hooker (Cyperaceae), is often dominant (Jarman et al. 1988). It is a significant landscape feature of western Tasmania occupying over 500 000 ha and has world heritage value (Balmer et al. 2004). In Tasmania, there have been calls to increase the amount of planned burning in buttongrass moorlands for the benefit of biodiversity in this habitat and to reduce the risk of catastrophic fires burning across the landscape into fire-sensitive vegetation communities or threatening human lives and property (Marsden-Smedley and Kirkpatrick 2000; Marsden-Smedley 2009). However, there has been very little investigation of the invertebrate fauna of buttongrass moorlands. Some opportunistic collections of invertebrates from this community have been summarised (Brown et al. 1993). A 10-day summer survey using sweep nets recorded 238 morphospecies primarily Araneae, Coleoptera and Diptera (Greenslade and Driessen 1999; Greenslade and Smith 1999). However, a number of groups were not identified including Hymenoptera, Lepidoptera and Acarina. During a 12-month survey of Collembola, using sweep nets and pitfall traps, nearly 30, 000 specimens representing 13 families and 42 taxa were collected (Driessen and Greenslade 2004). This study was a subset of a much larger collection of invertebrates which is the subject of this chapter. In a study using soil cores and sampling in four different seasons, over 12 000 Acarina representing 146 species from 72 families were collected (Green 2008, 2009); however a complete description of this fauna has not been published.

Here I describe invertebrate assemblages collected monthly over 12 months in buttongrass moorlands with different fire history at two locations. Specifically the following questions are addressed. (1) Do the invertebrate assemblages vary between months and are the monthly patterns of invertebrate occurrence the same between locations? (2) Do taxa that are common to both locations differ in their seasonal pattern of occurrence? (3) Do the invertebrate assemblages differ between old regrowth and young regrowth moorland?

2.3 Methods

Study sites.

The study was conducted at two locations (80 km apart) in western Tasmania; a lowland location (320 m asl) at McPartlan Pass (42° 51’ S 146° 12’ E) and a montane location (730–800 m asl) near Lake St Clair (42° 10’ S 146° 8’ E). Each location had two sites that had a buttongrass moorland plot that was burnt 4–11 years previously (young regrowth) immediately adjacent to a buttongrass moorland plot that was burnt 25–29 years previously (old regrowth). Sites were 7 km apart at the montane location and 3 km apart at the lowland location. Plots were a minimum of 40 m and a maximum of 80 m apart.

Both locations have similar mean annual rainfall (lowland, 1,951 mm; montane, 1,868 mm,) and monthly rainfall patterns with rainfall highest in winter and lowest in January–March. Minimum temperatures are 2–3°C lower throughout the year at the montane location (February, 7.3°C; July, –0.1°C) than at lowland location (February, 9.2°C; July, 3.2°C). During summer, maximum temperatures are similar at the two locations (19–20°C) but cooler during spring and autumn and up to 4°C cooler during winter at the montane location (6.2°C) than at the lowland location (10.0°C). At the montane location, the total rainfall for the period July 1999–June 2000 was 85% (1,591 mm) of the July–June average rainfall for the period 1990–2010 (1,868 mm). Mean maximum temperature for July 1999–June 2000 (14.0°C) was 0.9°C warmer than the average for the period 1991–2010. Mean minimum temperature for July 1999–June 2000 was 2.7°C was which the same as the long term average. At the lowland location, the total rainfall for the period July 1999–June 2000 (1,964 mm) was 13 mm over the July–June average rainfall for the period 1969–2010. Mean maximum and mean minimum temperatures for July 1999–June 2000 were 14.7°C and 6.6°C respectively, which were a little warmer than the long-term average (maximum, 14.0°C and minimum, 6.3°C).

The vegetation at each plot was assessed by recording the percentage cover of each plant species within a 2- by 2-m square centred on each of 10 pitfall trap locations (see below) and then averaged. Moorland sites at the lowland location were dominated by sedges, primarily *Gymnoschoenus sphaerocephalus* (plot

ground cover range: 23–42%), *Sporadanthus tasmanica* (Hooker) (Restionaceae) (2–8%), *Lepidosperma filiforme* Labillardière (Cyperaceae) (3–5%) and *Empodisma minus* (Hooker) (Restionaceae) (2–4%). There was a distinct shrub component at the lowland location comprised of *Leptospermum nitidum* Hooker (Myrtaceae) (3–10%), *Sprengelia incarnata* Smith (Ericaceae) (3–8%), *Bauera rubioides* Andrews (Cunoniaceae) (5–7%) *Boronia pilosa* Labillardière (Rutaceae) (3–5%), *Baeckea leptocaulis* Hooker (Myrtaceae) (1–4%) and *Epacris corymbiflora* Hooker (Ericaceae) (1–4%). Moorlands at the lowland sites are classified as "standard blanket moorland" (Jarman et al. 1988). This community is widespread and common in lowland areas of western Tasmania, occurring within a wide range of environmental situations on ancient sediments that carry shallow, acid and infertile soils (Jackson 1999).

At the montane location, the moorland sites were also dominated by sedges but to a greater extent than the lowland location*; Gymnoschoenus sphaerocephalus* (57–84%), *Sporadanthus tasmanica* (5–10%), *Lepidosperma filiforme* (3–6%) and *Empodisma minus* (0–10%). The only shrub recorded at the montane sites was *Sprengelia incarnata* (1–2%). Although a relatively small component of the community, grasses and herbs were more diverse and abundant at the montane location than the lowland location. The moorlands at the montane sites are classified as 'common highland sedgey' (Jarman et al. 1988). This community is widespread in highland regions of central and western Tasmania where it is associated with dolerite geology with relatively fertile soils (Jackson 1999).

There were more areas of bare ground and less litter and dead plant material at the lowland sites than at the montane sites. There were no clear differences in the floristics of the moorlands between old regrowth and young regrowth plots, but there was an observable difference in the vegetation structure. The older vegetation was taller and denser with more litter and dead plant material, while more bare ground was present among the younger vegetation.

Plate 9 Pitfall trap set in moorland

Plate 10 Sweep sampling in moorland

Invertebrate sampling.

All sites were surveyed once a month from July 1999 to June 2000 using sweep nets and pitfall traps (Plates 9–10). The sweep net comprised a 45.6-cm-diameter hoop, 71-cm handle and a white funnel tapered net that was 81 cm long with a 0.9- by 0.3-mm mesh size. A sweep sample comprised 100 passes of the net across the top of vegetation. One sweep of the net would occur at every pace of the collector. Two sweep samples were taken each survey and each sweep sample was performed in a random pattern within a 50-m radius of the pitfall trap lines described below. At the completion of the sample, all large twigs that were collected in the net were beaten against the inside of the net and removed. All invertebrates in the net were transferred into a screw-lidded jar (50 by 50 by 80 mm) containing 70% ethanol. Sweep samples were performed on clear days, when the vegetation was dry, between 10:00 a.m. and 6:00 p.m., and with maximum daily temperatures ranging from 9°C in July to 30°C in January. The exception to this protocol was in December at the lowland site when conditions were cooler than forecasted and the vegetation was a little damp from the previous night's rain.

A pitfall trap comprised a plastic 225-ml drinking cup with a 7-cm opening inserted into a PVC tube. It contained 100 ml of 70% ethanol. A wooden lid (15 by 15 by 0.3 cm) held up by wire legs was placed approximately 10 cm above the pitfall trap to prevent rain from entering. Ten pitfall traps were set in two lines of five traps with approximately 5 m between each trap and set for a period of seven days. Traps were not placed in hollows where they were likely to be flooded by surface run-off. All invertebrates were transferred on collection to a sample jar as above and topped up with 70% ethanol.

Invertebrates were initially sorted to major taxonomic groups (predominantly order) using a dissecting microscope at 10X magnification and the number of specimens were counted and stored in separate vials for each sweep and pitfall sample. Specimens within each taxonomic group were identified by specialists (listed in Appendix 1) to species or morphospecies (i.e. morphologically distinguishable) where possible and practical, and the numbers of each taxon were counted. The predominant collections of holometabolous and hemimetabolous insects were adults. Larvae of Lepidoptera, Diptera and Coleoptera were identified

and counted separately from adults even where adults and larvae of the same species were identified—because they have markedly different ecological roles. For taxa with similar-looking immatures and adults (e.g. arachnids, paurometabolous insects) these life stages were counted together. Immature Collembola, Hemiptera, Thysanoptera, Psocoptera usually could not be identified below family level and were counted separately.

Pitfall trapping and sweep net sampling contribute only a partial assessment of the entire insect population in buttongrass moorlands and the number of individuals and life stages caught by these methods reflects both activity and abundance of the sampled invertebrates. For convenience the term abundance is used here when referring to the number of invertebrates caught by these sampling methods.

Data analysis

Multivariate analyses of invertebrate assemblages used the software package PRIMER version 6.1 (Clarke and Gorley 2006) with the PERMANOVA + add-on (Anderson et al. 2008). The design consisted of four factors: Location (L; fixed with two levels: lowland and montane), Site (S; random with four levels and nested within Location), Age of Regrowth (A; fixed with two levels: old and young) and Time (T; fixed, 12 months, repeated measures), with n = 10 replicate observations per combination of factors for pitfall traps and n = 2 observations for sweep nets. Permutational distance-based multivariate analysis of variance (ANOVA), PERMANOVA (Anderson 2001; McArdle and Anderson 2001) was used to analyse the full design (1207 taxa X 96 samples), using 4,999 random permutations under a reduced model. All analyses were based on Bray–Curtis dissimilarities on square root transformed data. When there were too few possible permutations to obtain a reasonable test, a *P* value was calculated using 4,999 Monte Carlo draws from the appropriated asymptotic permutation distribution (Anderson and Robinson 2003). If, after PERMANOVA analysis, a term's component of variation was negative, the analyses was re-run pooling that term's contribution with other term(s) in the model which have equivalent expected mean squares. If more than one term had a negative error mean square then one term was removed at a time from the model

starting with the term having the smallest mean square (Anderson et al. 2008). To visualise relationships among samples, distances among centroids were calculated and ordinated using nonmetric multidimensional scaling (MDS) and clusters derived from agglomerative hierarchical clustering (group average linkage) were superimposed on the ordination to assist with showing relationships between samples.

Groups of taxa that had similar monthly distribution patterns were identified using agglomerative hierarchical clustering (Ward's linkage) in Minitab statistical software. The square of Euclidean distance/variance (= squared Pearson in Minitab) was used to standardise variances and to accentuate differences. Only taxa with a sample size ≥30 at a location were included in the cluster analysis. To characterise the monthly occurrence of each group of species the mean frequency distribution for each month (= monthly total/annual total X 100 averaged over all taxa in each group) was calculated. Seasonal lengths for each taxon were also calculated (Wolda 1979; Novotny and Basset 1998). The seasonal length is 12 months minus the longest number of monthly sampling periods in which the taxon was not observed.

Analysis of variance was used to compare temporal and spatial patterns of occurrence for taxa common to both locations. The model of analysis was the same as that described for the multivariate analysis above. The analysis was performed on log(x+1) transformed data in Minitab. Only taxa with a sample size ≥30 at each location were analysed. Taxa that could not be identified below family level and likely to comprise several species were not analysed; Hemiptera nymphs, Sterrnorrhyncha indet., Dicyrtomidae indet., Katiannidae indet. imm., Neanuridae indet. imm. and Odontellidae indet. imm.

2.4 Results

A total of 58,056 invertebrates were collected in pitfall traps and sweep nets during the 12 months of sampling representing 28 orders, 245 families and 1,207 taxa (= lowest recognisable taxonomic unit). Of the 1,207 taxa, 96% were recognised as species or morphospecies. Nearly half were identified to the level of either species (13%, 152 species) or genus (36%, 435 genera). Many taxa were collected only once (pitfall: 40%, n = 627; sweep: 42%, n = 826) and most were collected ten times or

less (pitfall: 78%; sweep: 79%). Only 246 (20%) of the 1,207 taxa were recorded in both sweep and pitfall samples, with 580 (48%) recorded only in sweep samples and 381 (32%) recorded only in pitfall samples. The most diverse invertebrate groups, in terms of number of species and morphospecies, were Diptera (300), Hymenoptera (262), Araneae (214), Coleoptera (105), Hemiptera (68), Acarina (66), Lepidoptera (61) Collembola (48) and Thysanoptera (20). The most abundant taxa were Collembola (28,167), Diptera (9,788), Araneae (4,147), Acarina (4,101), Hymenoptera (3,781), Hemiptera (2,900) and Orthoptera (2,241).

General trends

There was considerable variation between months and locations in the number of invertebrates and taxon richness for each of the dominant groups and for all invertebrates combined (Total) (Figs 2.1–2.2). For both pitfall and sweep samples, PERMANOVA confirmed significant differences in invertebrate assemblages between locations and among months, with the differences among months varying with location (Table 2.1). The invertebrate assemblages did not differ significantly between old and young regrowth moorland (Table 2.1). For pitfall samples the component of variation attributable to location was greater than the components of variation attributable to month and the month by location interaction. In contrast, for sweep samples the component of variation attributable to location was less than the components of variation attributable to month and the month by location interaction (Table 2.1). An MDS plot of the monthly centroids shows the nature of this interaction (Fig. 2.3). The lowland invertebrate assemblage was clearly distinguished from the montane invertebrate assemblage and at both locations there was a shift in composition between the winter months and summer months. However, the shift in composition differed between locations. The lowland December sweep assemblage is an outlier with very low captures of invertebrates probably caused by cool and moist conditions during sampling.

There were five clusters that broadly grouped taxa that were most commonly caught in summer, autumn, winter, early spring and late spring (Fig. 2.4). Group A had the largest number of taxa (59, Appendix 2) comprising 74 frequency distributions (i.e. 15 taxa occurred at both locations). Most Group A taxa (37 out of

59, 63%) had seasonal lengths ≥8 months and they were most abundant in collections during the warmer months, particularly January–March. Group A included a small number of taxa (10) that had seasonal lengths ≤4 months peaking in January or February. Group A was dominated (in terms of number of taxa) by Diptera (12), Araneae (10), Collembola (10), Acarina (8) and Hemiptera (7). Group B comprised 12 taxa with narrow abundance peaks centred on April and included taxa belonging to Diptera (4) and Araneae (3) (Appendix 2). Only four Group B taxa had seasonal lengths ≥8 months. Group C had the second largest number of taxa (35) comprising 39 frequency distributions and they were most abundant in collections during the winter months. Most Group C taxa (24 out of 35, 69%) had seasonal lengths ≥8 months and the group was dominated by Collembola (18 taxa), Diptera (7 taxa) and Araneae (6 taxa) (Appendix 2). Group D comprised 12 taxa (14 frequency distributions) that were most commonly recorded in early spring and most were Diptera (7 taxa) (Appendix 2). Seven Group D taxa (58%) had seasonal lengths ≥8 months. Group E had 25 taxa which were most commonly recorded in late spring and tended to have narrow abundance peaks, with only seven taxa (28%) having seasonal lengths ≥8 months. Group E taxa were also dominated by Diptera (9 taxa) (Appendix 2).

There were 29 taxa identified to genus, species or morphospecies that were common to both locations (Appendix 2). For most of these taxa (19) the monthly frequency distributions for each location were placed in the same cluster group. The frequency distributions of eight taxa were placed in chronologically adjacent groups. For two taxa, *Eriophora pustulosa* (Walckenaer) (Araneidae) and *Polykatianna* cf. *aurea* (Womersley) (Katiannidae) the monthly frequency distributions for each location were clustered into opposite groups; A (summer) and E (winter).

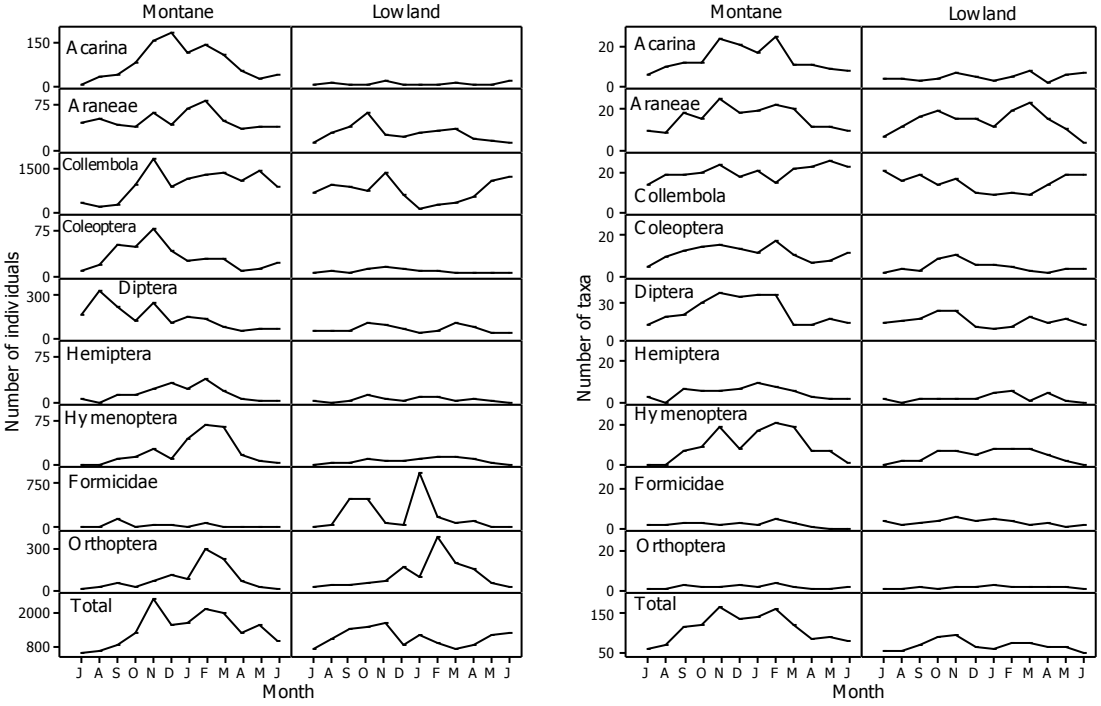

Fig. 2.1 Variation in monthly abundance and number of taxa in pitfall samples for each of the dominant orders of invertebrates and total invertebrates.

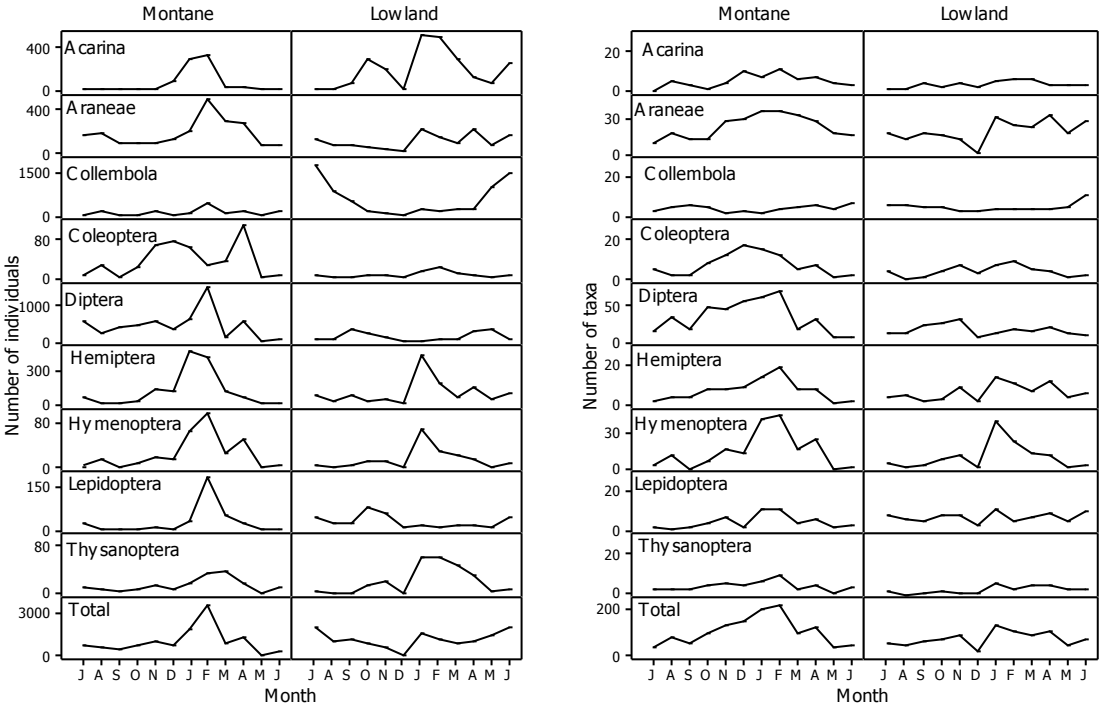

Fig. 2.2 Variation in monthly abundance and number of taxa in sweep samples for each of the dominant orders of invertebrates and total invertebrates.

Table 2.1 PERMANOVA of invertebrate assemblages obtained from pitfall traps and sweep nets showing the partitioning of multivariate variation and tests of Location, Site, Age of Regrowth, Month and their interactions. *P* values were obtained using permutations except where the number of possible permutations was not large and Monte Carlo *P* values were obtained (indicated by *). Sq. root = square root of the component of variation attributable to that factor in the model, in units of Bray–Curtis dissimilarity. Terms were pooled if their component of variation was negative.

Pitfall trap

Source	df	SS	MS	Pseudo-F	*P* (perm)	Sq. root
Location	1	3152.7	3152.7	5.7523	0.0376*	25.518
Age	1	242.4	242.4	1.0483	0.4108	1.6710
Site (Location)	2	1096.2	548.08	2.3703	0.0366	12.587
Pooled	3	693.68	231.23			15.206
Total	7	5185				
Month	11	68101	6191	2.9834	0.0002	22.682
Location x Month	11	39146	3558.7	1.7149	0.0002	19.259
Pooled	73	1.51E+05	2075.1			45.554
Total	95	2.59E+05				

Sweep net

Source	df	SS	MS	Pseudo-F	*P* (perm)	Sq. root
Location	1	2789	2789	5.6789	0.0354*	23.968
Site (Location)	2	982.23	491.11	2.6786	0.0406	12.405
Pooled	4	733.39	183.35			13.541
Total	7	4504.6				
Month	11	98552	8959.3	4.2238	0.0002	29.236
Location x Month	11	60805	5527.7	2.6060	0.0002	29.183
Pooled	73	1.55E+05	2121.2			46.056
Total	95	3.14E+05				

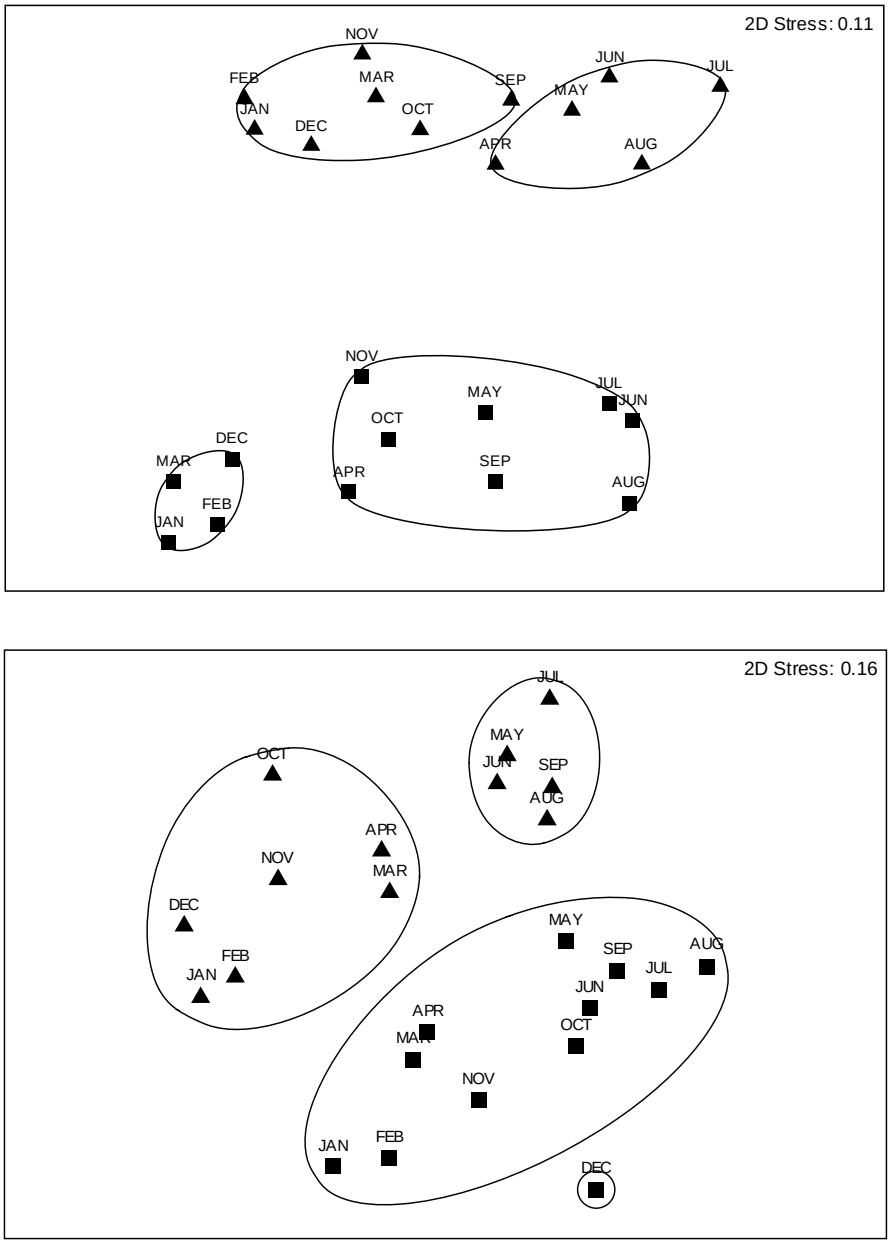

Fig. 2.3 MDS ordination of monthly centroids for pitfall trap (top) and sweep net (bottom) samples. Triangles, montane centroids; squares, lowland centroids.

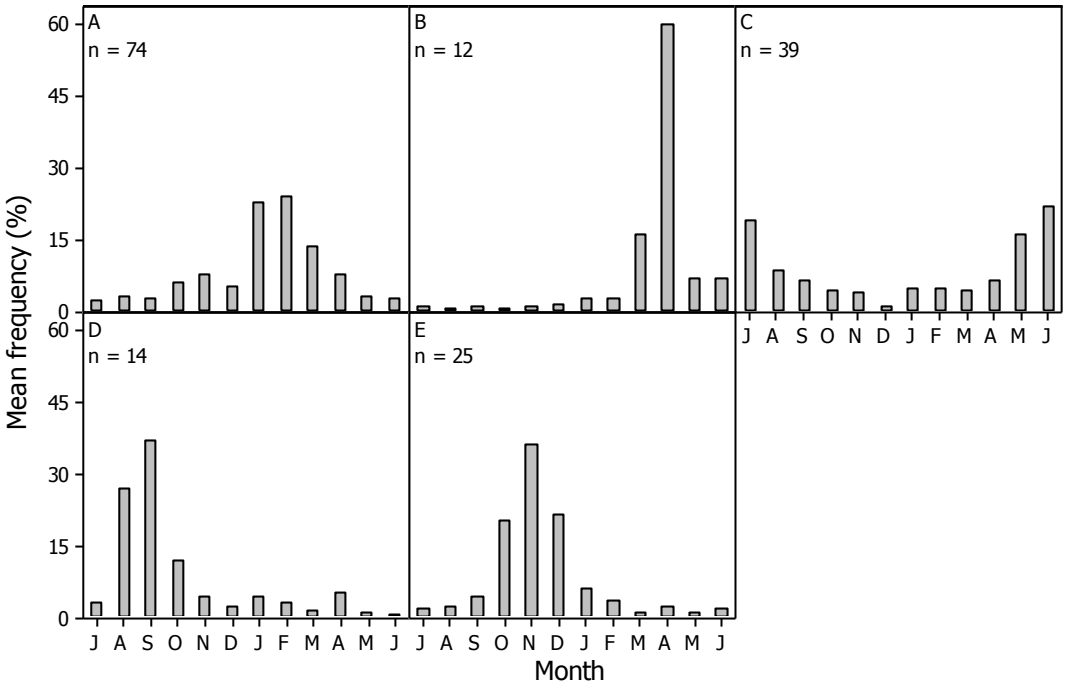

Fig. 2.4 Mean monthly frequency distributions for five groups of taxa (A–E) derived from cluster analysis (Ward's linkage). N = number of frequency distributions.

Differences between locations

The invertebrate assemblages differed markedly between the two locations with only 29% of the taxa in common. The number of taxa was 1.4 times greater in montane moorlands than in lowland moorlands, with the Acarina, Coleoptera, Diptera, Hymenoptera and Hemiptera more taxon rich at the montane location. These taxa were also more abundant in the montane moorlands except for Acarina in sweep samples (Figs 2.1–2.2). Formicidae and Collembola were more abundant in pitfalls and sweeps respectively in lowland moorlands than in montane moorlands (Figs 2.1–2.2).

Variability among months

At both locations and for both survey methods the winter invertebrate assemblages were most dissimilar from the summer invertebrate assemblages (Fig. 2.3). The winter invertebrate assemblages had lower taxon richness and lower invertebrate abundance, except in lowland moorlands in sweep samples where total invertebrate abundance was greatest in winter—primarily because of the great abundance of Collembola. The spring and autumn invertebrate assemblages were intermediate between summer and winter but also had their own characteristic taxa (Appendix 2).

Overall there was generally low similarity between monthly invertebrate assemblages with many taxa occurring for a limited number of months or, where present all year round, they peaked in abundance at certain times, often in summer (Fig. 2.4, Appendix 2). The similarity between monthly invertebrate assemblages varied for most months between locations. There was less similarity between monthly assemblages for sweeps samples than for pitfall samples in part because sweeps were taken on one day each month and not taken from precisely the same spot within a plot. In comparison, pitfall samples were set for a week and were reset on precisely the same spot each time.

In pitfall samples, the summer (December–March) invertebrate assemblage was more dissimilar from the autumn–spring assemblage at the lowland location than at the montane location (Fig. 2.3). This separation of summer lowland invertebrates was largely a result of a decrease in the abundance and diversity of

Collembola (particularly *Paronellides* sp. 1 (Paronellidae), *Polykatianna* cf. *aurea* (Womersley), *Parakatianna* sp. 1 (Katiannidae), *Acanthomurus* spp. (Isotomidae), and *Lasofinius* spp. (Tomoceridae)) and an increase in abundance of Formicidae (particularly *Anonychomyrma* nitidiceps? (E. André), *Iridomyrmex* sp. 1 (anceps group)) (Fig. 2.1). In contrast, at the montane location, there was no overall decrease in the abundance or diversity of Collembola or an increase in the abundance of Formicidae (Fig. 2.1).

In sweep samples, the mid-autumn–early-spring invertebrate assemblages were more dissimilar from the remaining monthly assemblages at the montane location than at the lowland location (Fig 2.3). This appears to be associated with a general low taxon richness and low invertebrate abundance at this time of year at the montane location compared with the lowland location (Figs 2.1–2.2). The greater dissimilarity between April–May and January–February invertebrate assemblages at the montane location is also associated with a general decrease in invertebrate richness and abundance (Fig. 2.2).

Monthly Variability in Taxa Common to Both Locations

There were no significant interactions between location and month for 11 (38%) of the 29 taxa common to both locations (Table 2.2). For the other 18 taxa, several showed a broad consistency between locations with peaks in abundance occurring within the same 2–3 months of each other (e.g. *Araneus* sp. 2 (Araneidae), *Pardosa* sp. 1 (Lycosidae), *Lasofinius* spp); however, for the remaining taxa there were large differences in monthly abundances between locations (Fig. 2.5).

There did not appear to be any consistent pattern of taxa peaking in abundance early or later at either location. Eleven taxa had their abundance peak in the same month at both locations, four peaked slightly earlier at the montane location, six peaked slightly earlier at the lowland location and ten were too difficult to determine.

For three taxa there were significant interaction effects between location, month and age of regrowth on abundance. Both *Wartookia rebeccae* Southcott (Erythraeidae) and *Lasofinius* spp. were more abundant in old regrowth than young regrowth moorland at the lowland location but not at the montane location. In

contrast *Bobilla poene* Otte and Alexander (Gryllidae) (Plates 11–12) was more abundant in young regrowth than old regrowth at the montane location and not at the lowland location.

Plate 11 Male swamp cricket *Bobilla poene* Otte and Alexander (Orthoptera: Gryllidae)

Plate 12 Female swamp cricket *Bobilla poene* Otte and Alexander (Orthoptera: Gryllidae)

Table 2.2 Number of taxa common to both locations with a significant ($P < 0.05$) interaction effect between location (L) and month (M). Taxa are grouped by taxonomic order. Error term for L X M test is M X Site (Location). Maximum degrees of freedom are 10 and 20 for L X M and error respectively.

Order	No. of taxa	No. of significant ($P < 0.05$) location X month interactions
Acarina	3	2
Araneae	6	6
Collembola	5	3
Diptera	10	6
Hemiptera	1	0
Orthoptera	1	0
Formicidae	1	0
Lepidoptera	1	0
Thysanoptera	1	1
Total	29	18

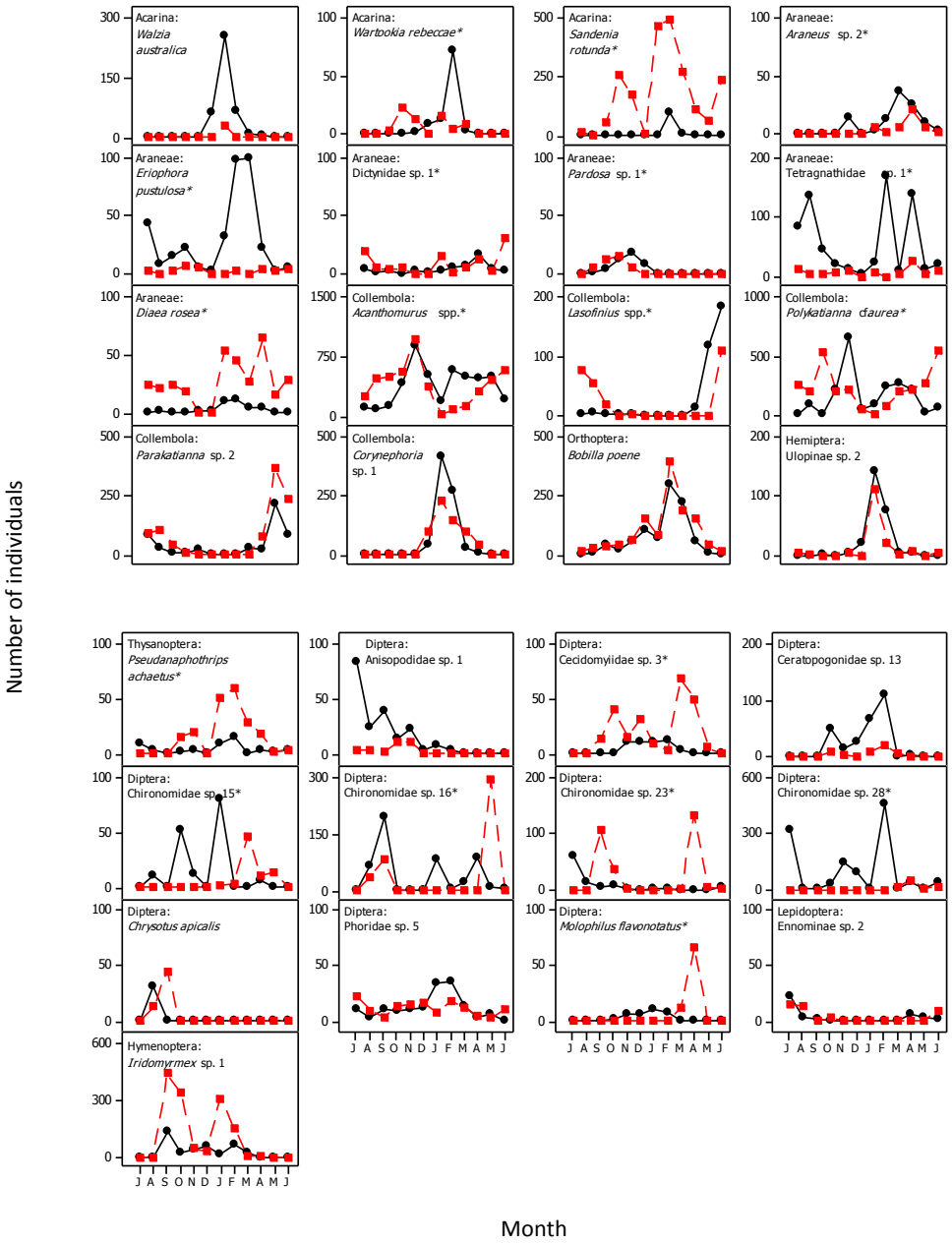

Fig. 2.5 Comparison of total monthly counts of taxa common to lowland (red squares) and montane (black circles) locations. * equals taxa with significant difference in monthly abundance patterns between locations using ANOVA.

2.5 Discussion

Monthly Variation in Invertebrate Community Assemblages

Monthly invertebrate assemblages in cool temperate buttongrass moorlands varied considerably throughout the year with groups of taxa occurring, or peaking in abundance, at different times of the year. However, at both locations and for each sampling technique, there was a general pattern of summer invertebrate assemblages being most dissimilar from the winter assemblages, and the spring and autumn assemblages being intermediate between the two. Taxon richness and abundance was generally higher during the warmer months than during the winter months, which is consistent with previous studies investigating invertebrate seasonality in temperate ecosystems (Nelson 1965; Nielsen 1974; New 1979; Lowman 1982; Abraham 1983; Baribeau and Maire 1983; Ohmart et al. 1983; Recher et al. 1983; Woinarski and Cullen 1984; Bell 1985; Masteller 1993; Recher et al. 1996; Gutiérrez and Menénendez 1998; Southwood et al. 2004; Cardoso et al. 2007; Chatzaki et al. 2009). However, large numbers of Collembola were found in lowland moorlands during winter which caused the total count of invertebrates in pitfall samples to be higher in winter than during summer. The montane moorlands also included Collembola taxa that were only recorded during the cooler months and some, such as species of the family Paronellidae, were only found at this location. Collembola can be extraordinarily cold-hardy and are very common in many arctic and antarctic habitats (Danks 1978). In a study of Collembola in riparian habitats in France, some species were active during winter but overall abundance peaked in summer (Lek-Ang and Deharveng 2002). In eucalypt forest canopies, Collembola were most abundant in winter in western Australian forests and autumn in eastern Australian forests (Recher et al. 1996). The increase in Collembola in forest canopies during these wetter seasons was suggested to be associated with maximum microbial and fungal growth. In addition to Collembola, several species of, Araneae, Diptera and Lepidoptera larvae were only recorded during winter in our study. Other studies have also recorded Araneae, Diptera and Lepidoptera larvae during winter in temperate ecosystems (Nielsen 1974; New 1979; Recher et al. 1996).

Although the overall separation between the winter and summer invertebrate assemblages was similar between locations, the shift in invertebrate assemblages between these seasons differed between locations. Compared with the lowland location, the montane invertebrate assemblages in the cooler months were more dissimilar from warmer months, and were characterised by low invertebrate abundance and richness. These differences between locations are probably largely driven by differences in temperature, which is a key driver for insect growth, development and reproduction (Masaki and Wipking 1994; Gullan and Cranston 2005). This separation between cool and warm month assemblages at the montane location is accentuated for invertebrate fauna sampled by sweep nets. This is consistent with temperature being an important driver for insect activity because invertebrates that are active among the tops of foliage are more exposed to adverse climatic conditions than the more sheltered ground-active fauna.

The lowland summer pitfall assemblages were more dissimilar from the remaining months compared with the montane moorland pitfall assemblages. This difference was primarily associated with a decrease in the richness and abundance of Collembola and an increase in the abundance of Formicidae in the lowland moorlands. The reason for this difference between locations is not clear but may also be driven by differences in temperature between the two locations. Availability of moist habitats and refuges are strong influences on Collembola assemblages (Hopkin 1997; Lek-Ang and Deharveng 2002) and Collembola have been recorded moving deeper into soil during summer to avoid adverse dry conditions (Hale 1966). The moorlands at the lowland location are probably more susceptible to drying out in summer than at the montane location because of higher ambient temperatures, more bare ground, and less litter and dead vegetation. The peats in lowland moorland often become dry, cracked and hard during summer (personal observation). Such conditions may also restrict the microbial and fungal growth on which many Collembola feed. Another possible reason for the decrease in Collembola during summer in lowland moorlands may be predation by Formicidae (Hopkin 1997; Ferguson 2004). During summer there was a significant increase in numbers of Formicidae at the lowland location which was not recorded at the montane location. Both these hypotheses require further investigation.

Although monthly invertebrate assemblages varied considerably, with groups of taxa peaking in abundance at different times of the year, further sampling over more years is required to determine if the patterns observed are consistent from year to year. Previous studies in temperate ecosystems have found that seasonal patterns can vary between years or remain largely the same. In a steppe ecosystem, spider assemblages showed consistent seasonal patterns during a two-year study (Weeks and Holtzer 2000). No significant changes in the seasonal pattern of occurrence of canopy invertebrates were found in oak trees over five years of monitoring although there were changes in the magnitude of abundance (Southwood et al. 2004). Similarly, in a three-year study in eucalypt forests a regular pattern of summer peaks and winter troughs of invertebrate biomass was found but the ratio between the summer peak and winter trough decreased during drought (Bell 1985). However, In the Mediterranean climate of Western Australia the variability in canopy arthropod numbers between years could be as great as that between seasons (Recher et al. 1996). The general patterns of insect occurrence reported here are not expected to vary considerably between years because western Tasmania has a distinct seasonal cycle and climate conditions during our survey were consistent with this seasonal cycle. However, annual rainfall during the year of survey at Lake St Clair was 15% less (277 mm) less than the long-term average (1,868 mm).

Monthly variation in taxa common to both locations

For most taxa common to both locations the monthly pattern of abundance was the same or, if they differed significantly, had the same general pattern displaced by a month or two. Several previous studies comparing seasonality in species occurring at different altitudes have found that at higher altitudes species become more active earlier in the season (Baribeau and Maire 1983; Gutiérrez and Menénendez 1998). There was no consistent pattern of species activity occurring earlier or later at our two study locations with many taxa having similar activity patterns at both locations. This suggests that the temperature difference between the two locations was not having a significant influence on the timing of activity at the montane location compared with the lowland location for many of the taxa that were

common to both locations. However, it may also be possible that the frequency of sampling used in the study (once a month) may have missed more subtle differences in the timing of activity of these taxa. There were also many taxa with monthly activity patterns that differed markedly between locations with no consistent pattern to this variation. These differences in monthly activity patterns between locations may be driven by localised factors. The specific timing of abundance and activity of invertebrate life stages during the warmer months can be influenced by local factors such as timing of new leaf growth, flowering, fruiting, rainfall events and microenvironment variation in moisture availability and solar radiation (Denlinger 1980; Lowman 1982; Badejo 1990; Danks 2006). Further research is required to determine the local influences on invertebrate activity in buttongrass moorlands.

Variation in invertebrate community composition between old and young regrowth moorland

The lack of evidence that the invertebrate assemblages differed between old (25–29 years) and young (4–11 years) regrowth sites at our two locations is consistent with the minimal variation in invertebrate species richness and abundance found in buttongrass moorlands ranging in post-fire age from 5 to 64 years (Greenslade and Driessen 1999). In a study of soil mites using soil cores, Acarina density and richness also did not differ significantly in moorland regrowth aged between <5 and 30 years (Green 2009). However, Acarina density and richness was significantly higher 30+ years post-fire compared with younger age classes, suggesting that Acarina populations may take a long time to re-establish following fire in buttongrass moorland (Green 2009). Although no differences in invertebrate assemblages were found between old and young regrowth, a small number of taxa appeared to be influenced by differences in age of regrowth. An earlier study on Collembola also found fire-age effects for individual taxa (Driessen and Greenslade 2004). Further research, using well designed studies, is needed to investigate invertebrate succession following fire in buttongrass moorland.

Implications for invertebrate surveys

The optimum time to conduct invertebrate surveys in buttongrass moorland, in terms of taxon richness and abundance, is mid- to late summer (January–February)—at least for the sampling methods used. For pitfall samples, invertebrate taxon richness and abundance was also high during late spring. Using the results of this 12-month survey, an invertebrate survey conducted in either January or February will account for around a third of the total number of taxa. Collembola were more abundant and diverse during the cooler months particularly in lowland moorlands. Management-focussed research—such as effects of fire on invertebrates—is frequently conducted during one or two months of the year (e.g. Greenslade and Driessen 1999; York 2000; Parr et al. 2004; Brennan et al. 2006), thus limiting the scope of conclusions, and this limitation is not always acknowledged.

2.6 Conclusions

Seasonal variation in diverse, multi-species invertebrate assemblages proved diverse in its patterning at both the assemblage and individual taxon levels. There was less asynchrony in abundance of taxa between locations than might have been expected from their environmental differences and the effects of the age of regrowth were not evident at the assemblage level and were rare among taxa. The conclusion that comprehensive sampling of temperate invertebrate biodiversity requires sampling in all seasons was strongly reinforced.

Chapter 3

Successional changes in moorland invertebrate assemblages following fire are predicted by changes in environmental variables: evidence from a space-for-time approach

3.1 Abstract

Fire management in protected areas requires an understanding of the consequences of fire regimes. Invertebrates are a key component of biological communities, but studies of fire impacts on diverse invertebrate assemblages over long timeframes are rare. The responses of ground- and foliage-active invertebrate assemblages to fire in buttongrass moorlands were investigated using a space-for-time design. Assemblages in recently burnt moorlands were distinct from those in older moorlands. Contrary to expectations, ground-active invertebrate abundance, but not taxon richness, was greatest in young regrowth (2–3 years since last fire), owing to large populations of Formicidae, Orthoptera, Collembola and Diptera. Foliage-active assemblages followed the expected trend with least numbers of invertebrates and taxa in young regrowth. Very few commonly caught taxa were absent from young successional stages and none were absent from later successional stages. Invertebrate assemblages in moorlands on low productivity soils took approximately twice as long to return to the pre-fire state than assemblages on moderate productivity soils. The shifts in invertebrate composition were associated with shifts in vegetation composition. Vegetation density was found to be a potentially important predictor of invertebrate compositional variation. Fire in buttongrass moorland appears to have a limited impact on ground-active and foliage-active invertebrate assemblages, suggesting that these components of the invertebrate fauna are resilient to fire.

Key words: arthropods, soil nutrients, buttongrass moorland, Tasmania, conservation

3.2 Introduction

Fire is widely used by land managers for ecosystem management (Whelan 1995; Keith et al. 2002b; Parr and Chown 2003). Managers of protected areas need to determine the most appropriate fire regime to satisfy a range of objectives, including biodiversity conservation and protection of life and property. Invertebrates comprise more than 75% of species and are critical for sustaining ecosystem processes (Kim 1993). Studies on the effects of fire on invertebrates are frequently constrained by the cost of managing and identifying large numbers of specimens, insufficient taxonomic knowledge and the availability of taxonomists. Consequently, most studies have been short-term, focussing on the immediate post-fire impacts, and have typically used either order-level identification for many groups or species-level identification for one or two invertebrate groups. Studies that involve long time-frames and that cover a wide variety of groups with species-level identification are rare, and include studies in American cottonwood forests (Bess et al. 2002), Swiss chestnut forests (Moretti et al. 2004) and South African fynbos (Pyrke and Samways 2012b). Information on the effects of fire on a broad range of invertebrate taxa is important for making appropriate decisions on fire management for optimal biodiversity conservation (Pyrke and Samways 2012b).

Invertebrate responses to fire tend to be variable, difficult to detect and with few consistent patterns owing to variation in level of identification, sampling methods, experimental designs, and fire regimes (Tscharntke and Greiler 1995; Swengel 2001; Whelan et al. 2002; Parr and Chown 2003; Joern and Laws 2013; New 2014). One generally consistent fire response pattern is that the compositions of invertebrate assemblages change immediately following fire with most taxa decreasing markedly in abundance (Warren et al. 1987; Friend 1995b; Whelan 1995; Swengel 2001; Whelan et al. 2002; New 2014). This decrease is often associated with post-fire changes in composition and structure of vegetation and other environmental elements (e.g. Swengel 2001; Moretti et al. 2002; Brennan et al. 2006; Underwood and Quinn 2010). Some invertebrate taxa appear to benefit from early post-fire conditions. Many studies have shown an increase in abundance in some Formicidae (O'Dowd and Gill 1984b; Andersen and Yen 1985; Bess et al. 2002; Niwa and Peck 2002; Greenslade and Smith 2010; Underwood and Quinn 2010;

Matsuda et al. 2011; Moranz et al. 2013) and Orthoptera (Friend and Williams 1996; Bess et al. 2002; Hochkirch and Adorf 2007; Greenslade and Smith 2010), although these increases in abundance may also be due to an increase in activity (Melbourne 1999; Gullan and Cranston 2010). Both long-term studies and studies comparing communities with a wide range of fire ages (e.g. space-for-time designs) have shown that as vegetation and other environmental elements return to pre-fire conditions the invertebrate assemblages often return to pre-fire levels of diversity, abundance and composition, particularly in fire-prone ecosystems (Neumann and Tolhurst 1991; Friend and Williams 1996; Andersen and Müller 2000; Bess et al. 2002; Parr et al. 2004; Munro et al. 2009; Cruz-Sánchez et al. 2011; Pyrke and Samways 2012b). A few studies involving several invertebrate groups have found no change in invertebrate abundance after fire (Abbott 1984; Sieman et al. 1997; Coleman and Rieske 2006).

Buttongrass moorland is a treeless sedgeland in which the hummock sedge, commonly known as buttongrass, *Gymnoschoenus sphaerocephalus* (Brown) Hooker (Cyperaceae) is often dominant (Jarman et al. 1988). Buttongrass moorland is a fire-prone and highly flammable vegetation type that is intermixed with fire-sensitive rainforest and alpine vegetation in western Tasmania. Buttongrass moorland was burnt by hunter-gatherers to facilitate easy passage and to encourage game (Thomas 1993; Marsden-Smedley 1998), perhaps extending the range it occupied in interglacial conditions before humans entered Tasmania. The displacement of indigenous people by Europeans in the nineteenth century appears to have increased fire incidence in inland areas, where extensive areas of rainforest have been burned (Marsden-Smedley 1998; Johnson and Marsden-Smedley 2001), but decreased fire incidence near the coast (di Folco and Kirkpatrick 2013). Increased burning in buttongrass moorlands has been advocated for the benefit of the species that depend on this habitat, and to reduce the risk of catastrophic landscape-scale fires burning into fire-sensitive vegetation (Marsden-Smedley 1998; Marsden-Smedley and Kirkpatrick 2000; Marsden-Smedley 2009). However there has been very little investigation of the response of invertebrate assemblages to fire to assist with the development of appropriate fire regimes (Driessen 2010).

The short (<5 years) and long term effects (up to 60 years) of burning on ground- and foliage-active invertebrates in buttongrass moorlands were investigated and potential predictors of invertebrate compositional change were identified. Because buttongrass moorland is a fire-prone vegetation type, I expected that the invertebrate community would be resilient to fire. I expected that fire-induced loss of environmental components (e.g. vegetation biomass, structural complexity, litter and decaying vegetation) would result in a short-term change in assemblage composition and a decrease in invertebrate abundance and species diversity. Formicidae and Orthoptera were expected to increase in abundance in the short-term following fire. As the environmental components returned to the pre-fire state, I expected that the invertebrate community would become increasingly similar to long-unburnt invertebrate communities.

3.3 Methods

Study sites

The study was conducted in lowland moorland near Lake Pedder (42° 57’ S, 146° 11’ E) and in montane moorland near Lake St Clair (42° 06’ S, 146° 11’ E). Although the altitude differs between locations (lowland, 320 m; montane, 730–800 m), both have similar mean annual rainfall (lowland, 1,951 mm; montane, 1,868 mm) and monthly rainfall patterns, with rainfall highest in winter and lowest in January–March. Minimum temperatures are 2–3°C lower throughout the year at the montane location (February, 7.3°C; July, –0.1°C) than at the lowland location (February, 9.2°C; July, 3.2°C). During summer, maximum temperatures are similar at the two locations (19–20°C) but cooler during spring and autumn and up to 4°C cooler during winter at the montane location (6.2°C) than at the lowland location (10.0°C).

Moorland vegetation at the two locations differs primarily because of differences in geology and soil fertility. Moorlands at the lowland location were dominated by sedges, primarily *Gymnoschoenus sphaerocephalus*, *Lepidosperma filiforme* Labill. Labillardière (Cyperaceae) *Sporadanthus tasmanica* (Hook. f.) B.G.Briggs & L.A.S.Johnson, *Empodisma minus* (Hook. f.) L.A.S.Johnson & D.F.Cutler and *Chordifex hookeri* (D.I.Morris) B.G.Briggs (Restionaceae), and also had a distinct shrub

component of *Leptospermum nitidum* Hook. f., *Melaleuca squamea* Labill., *Baeckea leptocaulis* Hook. f. (Myrtaceae), *Sprengelia incarnata* Sm., *Epacris corymbiflora* Hook. f. (Ericaceae), *Bauera rubioides* Andrews (Cunoniaceae) and *Boronia pilosa* Labill. (Rutaceae). This community is widespread and very common in lowland areas of western Tasmania, occurring within a wide range of environmental situations on siliceous rock types with shallow, infertile soils (Jarman et al. 1988). Moorlands at the montane location were dominated by sedges, particularly *Gymnoschoenus sphaerocephalus* and *Lepidosperma filiforme* but the shrub layer was very sparse with *Sprengelia incarnata* the most common species. Although grasses and herbs were a small component compared with sedges they were more common in these moorlands than those at the lowland location. The montane moorlands are widespread in highland regions of central Tasmania where they are associated with dolerite geology and relatively fertile soils (Jarman et al. 1988).

Experimental design

I used a space-for-time substitution design (Clarke 2008; Driscoll et al. 2010) comprising forty-four sites: 19 in lowland moorland and 25 in montane moorland (Fig. 3.1). Age of regrowth categories (Table 3.1) were established *a priori*. The original design had a minimum of five replicate sites within each age of regrowth category interspersed throughout each location with the following characteristics within each location: same geology; dominated by buttongrass, unburnt within 10 years prior to last burn; flat (less than 2 degrees of slope); similar altitude; and easy access (less than 0.5 hour walking distance from vehicle access) (Table 3.2, Fig. 3.1). This design was achieved for the montane location but at the lowland location there were few available moorland sites that were less than 30 years old since last fire and the interspersion of sites throughout the location was constrained. The two locations were analysed separately.

Sampling and identification of invertebrates

Pitfall traps and sweep nets were used to sample ground-active and foliage-active invertebrates respectively in late summer to maximise species richness and abundance (Chapter 2). Some components of the invertebrate community can only

be collected at other times of the year, thus limiting the scope of conclusions (Chapter 2).

At each site, seven pitfall traps were set, with one central trap and three 40-m 'arms' comprising two traps, with a 120° angle between arms. The distance between traps within an arm was 20 m. Traps were placed in gaps between buttongrass plants avoiding positions vulnerable to flooding (e.g. depressions). One month prior to survey, pitfall trap holes were dug using a 75-mm augur, a PVC tube (75 mm diameter by 110 mm depth) was inserted into the hole, and a 225-ml drinking cup with a lid was inserted into the tube. Pitfall traps were set at each site for two 14-day periods. At the montane location pitfall traps were set on 16–17th February 2004, cleared and reset on the 1st –2nd March 2004 and cleared on 15–16th March 2004. At the lowland location, pitfall traps were set on 19–20th March 2004, cleared and reset on 4–5th March 2004 and cleared on 18–19th March 2004. Pitfall traps contained 100 ml of 70% ethanol and a dash of glycerol. A temporary clear plastic roof was placed above the trap to help prevent flooding by rain.

A standard sweep sample consisted of 100 passes of the net across the top of the vegetation. The sweep net had a 45.6-cm diameter hoop, a 71-cm handle and a white funnel tapered net that was 81 cm long with a 0.9- by 0.3-mm mesh size. Three sweep samples were taken per site in the three sectors defined by pitfall trap layout. All material was transferred directly into a square jar (50 mm by 50 mm by 80 mm) with a screw lid containing 70% ethanol. All sweeps were performed between 10:00 a.m. and 18:30 p.m. on fine days, with temperatures ranging between 18 and 25°C. Because of the time required to survey all sites, sweep samples were conducted over a period of two days at each location (lowland: 24–25 February 2004; montane: 16–17 February 2004). To minimise any sampling bias the order of sweep sampling was stratified so that buttongrass moorlands of different age since last fire and in different areas of each location were allocated as evenly as possible on each day and at different times of the same day.

Invertebrates were initially sorted to major taxonomic groups (predominantly order) using a dissecting microscope at 10X magnification, and the number of specimens were counted and stored in separate vials for each pitfall and sweep sample. The majority of taxonomic groups were identified by specialists (listed in

Appendix 1) to species or morphospecies (i.e. morphologically distinguishable) where possible and practical, and the numbers of each taxon counted. Diptera and non-Formicidae Hymenoptera were identified to family level and the Psocoptera and Neuroptera were not identified beyond order level due to the loss of specimens. The predominant collections of holometabolous and hemimetabolous insects were adults. Larvae of Lepidoptera, Diptera, and Coleoptera were identified and counted separately from adults even where adults and larvae of the same species were identified—because they have markedly different ecological roles. For taxa with similar-looking immatures and adults (e.g. arachnids, paurometabolous insects) these life stages were counted together. Immature Collembola, Hemiptera and Thysanoptera usually could not be identified below family level and were counted separately.

Pitfall trapping and sweep net sampling contribute only a partial assessment of the entire insect population in buttongrass moorlands, and the number of individuals and life stages caught by these methods reflects both activity and abundance of the sampled invertebrates. For convenience, the term abundance is used here when referring to the number of invertebrates caught by these sampling methods.

Environmental variables

The area covered by plant species within a 1- by 1-m quadrat was recorded at each pitfall trap station. Most vascular plants were identified to species and non-vascular plant species were identified as far as practicable. For the purposes of analyses, plant species were pooled into the following life-form groups: shrubs, grasses, buttongrass, sedges (including rushes, but excluding buttongrass), herbs, cryptograms and ferns. The area covered by bare ground, litter, thatch, rock, pools, decaying buttongrass (= dead buttongrass) and decaying sedges excluding buttongrass (= dead sedge) were also recorded.

An index of vegetation density was recorded at each pitfall trap position at each site using a 100-cm-tall by 50-cm-wide board marked with 200 5- by 5-cm squares (Plate 13). The board was placed on the ground behind each pitfall trap and the number of squares obscured by vegetation was recorded by an observer

crouched five metres away. For each trap position vegetation height was recorded as the tallest height that had at least five squares in a row obscured by vegetation. Vegetation density and vegetation height were highly correlated ($R^2 > 0.99$); thus, only vegetation density was used in analyses. Peat depth (cm) was recorded next to each pitfall trap using a rod, which was inserted into the ground until it made contact with the underlying gravels. For each of the environmental variables measured at the seven pitfall traps the mean value was calculated and used in the analyses. Total area of buttongrass moorland and distance to nearest forest were measured from vegetation layers obtained from Tasmanian Vegetation Mapping Program (TVMP 2004).

Soil samples were collected on the 12th and 20th May 2004 at the lowland and montane locations respectively between 11:00 a.m. and 16:00 p.m. Four soil samples were collected adjacent (1–3 m) to each of the four inner most pitfall traps at each site. Samples were taken from between tussocks of buttongrass plants, avoiding pools of water. Each soil sample was taken using a 110-mm-long by 70-mm-diameter PVC tube that was inserted completely into the peat and then removed with the aid of a spade. The soil sample was retained in the PVC tube, sealed in a zip-lock plastic bag and kept cool for 24 hours. The top 5 mm of the sample was sliced off each sample to remove any surface vegetation. The samples were weighed, oven dried for 62 hours at 80°C and then reweighed to determine the percentage moisture content. The four samples from each site were then aggregated, put through a 1-mm sieve, had plant roots removed, and then analysed for 12 other soil variables: total nitrogen (%, Kjeldahl method), organic matter (%, loss on ignition), pH (1 part soil : 5 parts distilled water, shaken for 60 minutes), conductivity (µs/cm, 1 part soil : 5 parts distilled water, shaken for 60 minutes), available phosphorous (ppm, 1 part soil : 20 parts 1N sodium bicarbonate @ pH 8 (Olsen), shaken 30 minutes), Potassium, Calcium, Magnesium, Manganese, Zinc, Copper and Boron (ppm, 1 part soil : 5 parts 1N ammonium acetate @ pH 4.8, shaken 30 minutes).

The age of regrowth (= time since last fire) and number of fires that occurred since 1933 (fire frequency) were determined using a combination of fire history maps (maintained by the Tasmanian Parks and Wildlife Service), node counts on

Banksia marginata Cavanilles (Proteaceae) plants (Wills 2003), tree rings counts on Myrtaceae species and local knowledge. Almost all recent fires at study sites were planned burns conducted by the Tasmanian Parks and Wildlife Service during spring and autumn with fire intensities less than 500 kW m^{-1}. However, all sites have been subject to more intense summer wildfires since European settlement in Tasmania with major fires occurring in 1897/98, 1933/34, 1950s and 1972 (Marsden-Smedley 1998; Johnson and Marsden-Smedley 2001).

Plate 13 Board used to obtain an index of vegetation density

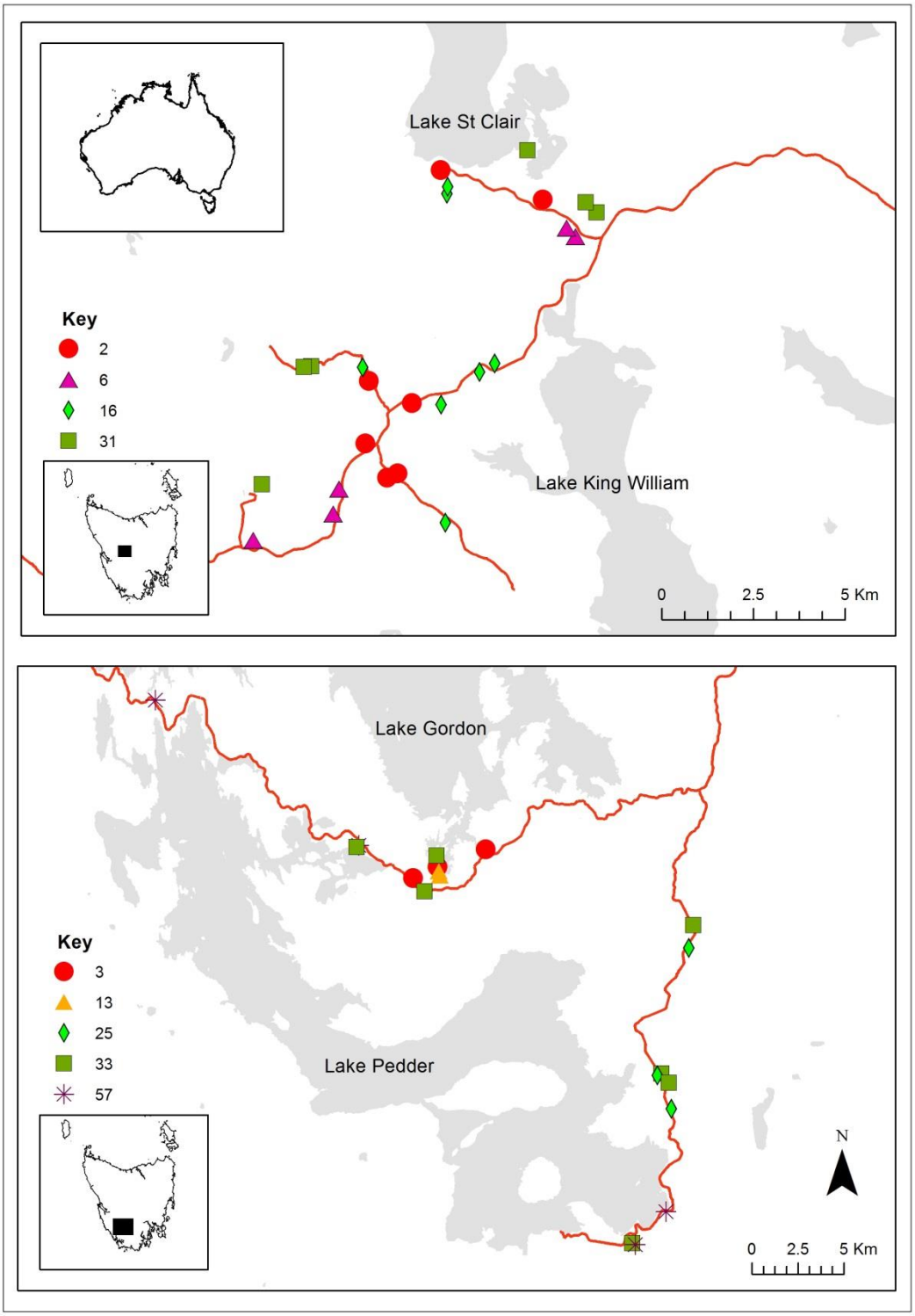

Fig. 3.1 Location of study sites in lowland moorland (bottom) and montane moorland (top). Numbers given in the keys are the mean ages (years) of the regrowth categories for sites. Shading indicates water bodies, solid line = roads.

Table 3.1 Age range and mean age (years) for, and number of sites (n) in, each regrowth category for lowland and montane moorland study locations.

Category	Lowland			Montane		
	Age Range	Mean Age	n	Age Range	Mean Age	n
1	3–3	3	3	1–3	2	7
2	-	-	-	5–8	6	5
3	12–13	13	2	13–20	16	7
4	22–27	25	3	-	-	-
5	31–40	33	7	30–31	31	6
6	54–65	57	4	-	-	-

Table 3.2 Fire history of survey sites in lowland and montane moorland locations between 1933 and 2004.

Lowland				
Site code	Time since last fire (yr)	Year of known fires	Number of known fires	Fire interval
AIRE	32	1972, 1950, 1933	3	19.5
AIRO	54	1950, 1933	2	17.0
CONN	31	1972, 1950, 1933	3	19.5
CONW	27	1977, 1950, 1933	3	22.0
COSE	25	1978, 1950, 1933	3	22.5
EDGR	54	1950, 1933	2	17.0
GECK	33	1971, 1950, 1933	3	19.0
MCPC	3	2001, 1972, 1933	3	34.0
MCPE	3	2001, 1988, 1972, 1933	4	22.7
MCPW	3	2001, 1972, 1933	3	34.0
MTAN	40	1950, 1933	2	17.0
RESA	12	1992, 1972, 1933	3	29.5
RESB	13	1991, 1972, 1933	3	29.0
SAND	22	1982, 1971, 1950, 1933	4	16.3
SENT	32	1972, 1933	2	39.0
SPUR	65	1933	1	-
STWN	54	1950, 1933	2	17.0
STWS	32	1972, 1950, 1933	3	19.5
WEDG	31	1972, 1933	2	39.0

Montane				
Site code	Time since last fire (yr)	Year of known fires	Number of known fires	Fire interval
BEDL	14	1990, 1933	2	57.0
BEEN	30	1974, 1933	2	41.0
BEES	5	1999, 1987, 1979, 1933	4	35.0
BURN	5	1999, 1984, 1979, 1933	4	22.0
COAT	17	1987, 1980, 1933	3	27.0
FLCK	14	1990, 1980, 1933	3	28.5
HARN	1	2003, 1989, 1979, 1933	4	23.3
HARS	1	2003, 1989, 1979, 1933	4	23.3
KWCE	3	2001, 1987, 1979, 1933	4	22.7
KWCW	5	1999, 1987, 1979, 1933	4	22.0
KWPL	13	1991, 1979, 1933	3	29.0
NAVR	3	2001, 1933	2	68.0
NPLE	16	1988, 1933	2	55.0
NPLW	3	2001, 1988, 1933	3	34.0
PUMP	31	1973	1	-
RCRE	17	1987, 1933	2	54.0
RCRN	30	1974, 1933	2	41.0
RCRW	30	1974, 1933	2	41.0
SCRN	2	2002, 1974	2	28.0
SCRS	8	1996	1	-
SCRW	8	1996	1	-

Lowland					Montane				
Site code	Time since last fire (yr)	Year of known fires	Number of known fires	Fire interval	Site code	Time since last fire (yr)	Year of known fires	Number of known fires	Fire interval
					SEPO	20	1984, 1933	2	51.0
					TREE	31	1973	1	-
					TREW	31	1973	1	-
					WEST	2	2002	1	-

Data analyses

Invertebrates

All multivariate analyses were carried out using the PRIMER version 6.1 computer program (Clarke and Gorley 2006) with the add-on package PERMANOVA+ (Anderson et al. 2008).

Invertebrate abundance data were log(x+1) transformed to down-weight the contributions of quantitatively dominant species (Clarke and Gorley 2006) and used to generate Bray–Curtis similarity measures. One-way analyses of similarity (ANOSIM) using 4,999 permutations were performed on the data to test for differences in the invertebrate assemblage composition between age of regrowth categories. ANOSIM returns an *R*-statistic which gives a measure of similarity between categories. R-values most commonly range from 0 to 1; the closer the *R*-value is to 1 the more different the categories are, while a value close to zero indicates assemblages can barely be separated (Clarke and Warwick 2001). As suggested by Clarke and Gorley (2006), large pairwise *R* values were used to indicate major differences between categories provided the global test was significant. ANOSIM tests are robust to differences in the number of samples in factor groups (Clarke and Warwick 2001). To visualise multivariate patterns in invertebrate assemblage structure, multidimensional scaling (MDS) was performed. The SIMPER routine was used to identify taxa that were important in discriminating between regrowth categories.

One-way analysis of variance was used to test the effect of age of regrowth on total invertebrate abundance, total taxon richness and abundance and taxon richness of numerically dominant (> 100 specimens, > 20 taxa) invertebrate groups.

Environmental variables

One-way analysis of variance was used to test the effect of age of regrowth on vegetation density, peat depth, total area of buttongrass and distance to nearest forest. ANOSIM using 4,999 permutations was used to test for the effect of age of regrowth on vegetation cover composition. Vegetation cover variables were square root transformed prior to analyses. Relationships among sites were visualised using MDS.

The RELATE routine was used to test if the among-sample relationships for invertebrate assemblages agree in the same way as among-sample relationships for vegetation cover composition.

Environmental predictors of invertebrate assemblages

The distance-based linear model (DISTLM) routine (Anderson et al. 2008) was used to model the relationship between the invertebrate assemblage structure and environmental variables. Prior to analyses, variables were either square root or natural log transformed based on examination of draftsman's plots. The number of variables was reduced to ensure there was a reasonable number of variables for the size of the data set (Anderson and Burnhan 2002). Principal components analyses were used to reduce the 13 vegetation and 13 soil variables down to two vegetation and two soil variables each for lowland and montane locations (Tables 3.3–3.4). From the remaining ten variables (Veg. PC1, Veg. PC2, Soil PC1, Soil PC2, vegetation density, number of fires, age of regrowth, peat depth, total area of moorland, and distance to nearest forest), Veg. PC1, vegetation height, total vegetation cover and age of regrowth were removed from the analysis because they were highly correlated ($[r] \geq 0.8$) with vegetation density at both locations. Marginal tests were performed to assess the statistical significance and percentage contribution of each environmental variable alone. Models were constructed from all possible combinations of the eight environmental variables using the BEST selection procedure. The corrected Akaike Information Criterion (AIC_c) was used to rank the models. The model with the lowest AIC_c value ($AIC_{c(min)}$) is the estimated 'best' of the candidate models. Because models within seven AIC_c units of $AIC_{c(min)}$ have some support and should rarely be dismissed (Burnham et al. 2011), the relative importance of predictor variables were assessed and ranked by summing AIC_c model weights across all models that included that variable (Burnham and Anderson 2002). The summed AIC_c weights for each variable can be interpreted as equivalent to the probability that the variable is a component of the best model (Symonds and Moussalli 2011). Predictor variables chosen in a model should not be interpreted as being necessarily causative as they may be acting as proxies for other

important variables that either were not measured or were omitted from the model for reasons of parsimony (Anderson et al. 2008).

The BIOENV routine was used to detect any variables that the linear DISTLM approach may have missed (Clarke and Gorley 2006; Hallet et al. 2012). BIOENV is a less constrained, fully non-parametric method which caters for non-linear functions (Clarke and Ainsworth 1993) and was used to find the best matches between the among-sample patterns for the invertebrate assemblage patterns and those from the environmental variables associated with those samples (Clarke and Gorley 2006). The null hypothesis of no similarities in rank order pattern between the complementary matrices was rejected if the significance level (*P*) associated with the test statistic (Spearman's rank 'matrix correlation' coefficient [ρ]) was <0.05. The extent of any significant differences was determined by the magnitude of ρ, with values close to zero indicating little correlation in rank order pattern whereas values close to 1 indicated a near perfect agreement. R^2 and ρ are broadly comparable since the latter is a matrix correlation, not a direct correlation (Hallet et al. 2012).

3.4 Results

Invertebrate abundance and richness

A total 83,272 ground- and foliage-active invertebrates comprising 290 families and 889 taxa were collected during February–March from lowland and montane moorlands. Slightly more taxa were recorded in pitfall samples (566, 64%) than sweep samples (534, 60%) although sweep samples contributed more specimens (46,603, 56%). Less than a quarter of all taxa recorded (211, 24%) were common to both pitfall and sweep samples indicating that the two methods were sampling substantially different sets of taxa. Similar numbers of taxa were recorded in lowland moorland (613, 69%) and montane moorland (617, 69%). The two moorland locations shared almost two thirds of the total number of taxa (558, 62%). More specimens were collected in lowland moorland (45,998, 55%) than in the montane moorland (37,274, 45%), despite less sample sites at the former (19 vs 25).

The most abundant invertebrate groups in decreasing order were: Collembola (27,293), Diptera (17,992), Acarina (10,173), Hemiptera (6,651), Araneae (5,327), Hymenoptera (3,897), Thysanoptera (3,443), Orthoptera (3,433), and Coleoptera (2,436). The groups with the most number of species/morphospecies were: Araneae (189), Coleoptera (147), Hemiptera (128), and Acarina (127); however, Diptera (41 families) and Hymenoptera (29 families) were identified to family level only and have been previously found to be more speciose than the other groups in buttongrass moorland (Chapter 2).

Changes in total invertebrate abundance and taxon richness with age of regrowth

The effect of age of regrowth on both total invertebrate abundance and taxon richness varied between locations and between sampling methods (Table 3.5). In lowland moorland, more ground-active invertebrates were captured in the youngest regrowth category (3 years) than in older categories owing to the large numbers of Collembola, Orthoptera, Diptera and Formicidae. Amphipoda were captured in greater numbers in the oldest regrowth categories. In contrast, less foliage-active invertebrates were captured in the youngest regrowth category, owing mainly to small numbers of Acarina, Hemiptera and Thysanoptera. Age of regrowth did not affect total taxon richness for ground-active invertebrates but did affect total taxon richness for foliage-active invertebrates. The number of foliage-active taxa was lower in the youngest regrowth category than in the older regrowth categories, with taxon richness for Acarina and Coleoptera lowest in the youngest age of regrowth.

In montane moorlands there was no effect of age of regrowth on total abundance and total taxon richness for ground- and foliage-active taxa; however, the directional trends were the same as in lowland moorlands (Table 3.5). There were larger numbers of Orthoptera and Formicidae and less Amphipoda in pitfall traps in the youngest age of regrowth in montane moorlands (Table 3.5). For foliage-active invertebrates, Acarina abundance and taxon richness were affected by age of regrowth with least numbers and taxa in the youngest regrowth. Counts of ground-active Hemiptera and Hymenoptera (excluding Formicidae) were higher in the 5–8 year regrowth compared with most other regrowth categories.

Table 3.3 Principal components for soil variables from sites in lowland and montane moorlands. Only variables contributing ≥ 0.25 to each factor are shown.

Factor	Variables with eigenvectors ≥ 0.25	Variation explained	
		%	Cumm. %
Lowland			
Soil PC1*	0.34 conductivity + 0.34 P + 0.34 K + 0.33 Mg + 0.32 N + 0.28 Zn + 0.26 organic matter + 0.26 Ca	52	52
Soil PC2*	– 0.45 Cu + 0.43 Ca – 0.37 Zn + 0.33 Mg + 0.31 N	15	67
Soil PC3	– 0.72 B – 0.46 Mn – 0.25 Cu	9	76
Soil PC4	–0.63 pH – 0.54 Mn – 0.25 Zn – 0.25 organic matter	8	84
Montane			
Soil PC1*	0.35 Mg + 0.35 conductivity + 0.32 soil moisture + 0.31 organic matter + 0.31 K + 0.60 Zn + 0.29 Ca + 0.28 P + 0.26 Mn + 0.25 N	53	53
Soil PC2*	– 0.41 Mn + 0.40 pH + 0.36 B – 0.30 organic matter + 0.29 Cu – 0.29 P – 0.26 N – 0.25 Ca	19	72
Soil PC3	0.63 Cu + 0.41 Zn – 0.32 Ca – 0.32 pH	9	81
Soil PC4	–0.67 B – 0.46 N + 0.37 P + 0.25 Ca	8	89

*principal components used to identify environmental predictors

Table 3.4 Principal components for vegetation cover from sites in lowland and montane moorlands. Only variables contributing ≥ 0.25 to each factor are shown.

Factor	Variables with eigenvectors ≥ 0.25	Variation explained	
		%	Cumm. %
Lowland			
Veg. PC1*	0.75 bare ground – 0.38 dead sedges – 0.35 dead buttongrass – 0.30 shrubs – 0.25 buttongrass	54	54
Veg. PC2*	0.58 buttongrass – 0.54 shrubs – 0.39 sedges – 0.34 cryptograms	23	77
Veg. PC3	– 0.85 cryptograms + 0.32 ferns	7	84
Veg. PC4	0.84 ferns – 0.33 litter	6	90
Montane			
Veg. PC1*	0.52 thatch + 0.44 bare ground – 0.39 dead sedges – 0.38 dead buttongrass – 0.38 buttongrass	47	47
Veg. PC2*	0.67 sedges –0.45 litter – 0.31 buttongrass + 0.27 shrubs + 0.27 thatch	22	69
Veg. PC3	0.59 dead sedges + 0.55 thatch + 0.27 cryptograms – 0.33 dead buttongrass – 0.27 cryptograms	10	79
Veg. PC4	–0.46 ferns + 0.45 dead sedges + 0.43 bare ground + 0.42 cryptograms – 0.27 buttongrass	7	86

Table 3.5 Mean (±se) abundance and mean (±se) taxon richness per pitfall trap and per sweep sample for invertebrate groups for each age of regrowth category at each moorland location. Only numerically dominant invertebrate taxa are shown. Significant effects determined using one-way ANOVAs; *P < 0.05, **P < 0.01, ***P < 0.001. Categories that do not share the same letters are significantly different from each other based on pairwise comparisons using Fisher's least significant difference method. N = number of specimens or number of taxa (note Diptera and non-Formicidae Hymenoptera identified only to family level).

Group	Lowland						Montane					
	n	mean age of regrowth (year)				P	n	mean age of regrowth (year)				P
		3	13	25	33	57			2	6	16	31	
Abundance – ground-active invertebrates													
All	17369	117.98 ± 7.45^{A}	89.25 ± 6.53^{B}	59.38 ± 4.97^{B}	57.00 ± 6.94^{B}	55.29 ± 7.96^{B}	***	19300	77.40 ± 18.97	63.37 ± 8.08	43.43 ± 6.72	64.02 ± 7.05	
Amphipoda	109	0.02 ± 0.02^{A}	0.04 ± 0.04^{AB}	0.41 ± 0.13^{B}	0.31 ± 0.15^{B}	0.91 ± 0.21^{B}	*	381	0.45 ± 0.13^{A}	0.78 ± 0.20^{AB}	1.66 ± 0.49^{B}	1.47 ± 0.37^{B}	*
Araneae	743	3.31 ± 0.06	2.21 ± 0.43	2.69 ± 0.60	2.67 ± 0.32	2.97 ± 0.28		1064	3.49 ± 0.57	2.82 ± 0.47	3.14 ± 0.70	2.82 ± 0.75	
Acarina	692	1.71 ± 0.62	4.00 ± 1.86	3.12 ± 0.70	3.03 ± 0.75	1.84 ± 0.5		1112	2.69 ± 0.71	3.87 ± 0.81	2.39 ± 0.52	4.26 ± 0.93	
Collembola	11614	80.76 ± 5.19^{A}	54.82 ± 9.18^{B}	37.52 ± 2.00^{B}	34.88 ± 6.32^{B}	32.03 ± 6.58^{B}	**	9712	38.9 ± 11.00	27.51 ± 6.06	16.46 ± 3.53	30.48 ± 6.96	
Orthoptera	927	10.48 ± 2.88^{A}	4.75 ± 0.96^{B}	1.29 ± 0.38^{B}	2.31 ± 0.64^{B}	1.54 ± 0.78^{B}	**	2281	13.49 ± 4.58^{A}	4.89 ± 1.62^{AB}	2.78 ± 0.83^{B}	4.45 ± 1.80^{B}	*
Hemiptera	262	1.14 ± 0.61	0.25 ± 0.11	0.29 ± 0.14	0.24 ± 0.10	0.63 ± 0.11		599	1.70 ± 0.45^{A}	3.08 ± 0.78^{B}	1.26 ± 0.22^{A}	1.21 ± 0.36^{A}	*
Coleoptera	156	0.67 ± 0.19	0.68 ± 0.25	0.45 ± 0.10	0.36 ± 0.11	0.86 ± 0.24		366	0.82 ± 0.18	0.81 ± 0.21	0.94 ± 0.27	1.69 ± 0.44	
Diptera	1423	8.95 ± 0.56^{A}	6.75 ± 1.18^{B}	4.60 ± 1.04^{C}	4.70 ± 0.32^{C}	3.97 ± 0.11^{C}	***	1730	5.38 ± 1.69	4.26 ± 0.86	4.62 ± 0.77	5.58 ± 0.61	
Hymen. ex. Formicidae	418	1.74 ± 0.51	1.46 ± 0.39	0.91 ± 0.27	1.57 ± 0.29	1.93 ± 0.34		762	1.11 ± 0.28^{A}	4.60 ± 1.77^{B}	1.64 ± 0.25^{A}	2.18 ± 0.78^{AB}	*

Group	Lowland						Montane					
	n	mean age of regrowth (year)				*P*	n	mean age of regrowth (year)				*P*
		3	13	25	33	57		2	6	16	31	
Formicidae	419	2.50 ± 0.07^{AB}	3.25 ± 0.89^{A}	1.57 ± 0.79^{BC}	0.99 ± 0.22^{C}	1.07 ± 0.18^{C}	**	508	2.89 ± 0.97^{A}	1.69 ± 0.44^{AB}	0.65 ± 0.13^{B}	0.56 ± 0.25^{B}	*
Taxon richness – ground-active invertebrates													
All	369	11.76 ± 0.13	11.57 ± 1.07	13.14 ± 1.26	11.27 ± 0.70	13.16 ± 1.17		436	14.01 ± 1.73	15.05 ± 1.59	13.56 ± 1.27	15.12 ± 1.33	
Araneae	76	2.55 ± 0.13	1.89 ± 0.18	2.07 ± 0.33	2.03 ± 0.15	2.14 ± 0.22		90	2.04 ± 0.19	1.77 ± 0.16	1.74 ± 0.15	1.81 ± 0.31	
Acarina	65	1.41 ± 0.49	2.04 ± 0.32	2.31 ± 0.46	1.89 ± 0.32	1.50 ± 0.34		76	1.79 ± 0.41	2.01 ± 0.30	1.65 ± 0.31	2.12 ± 0.25	
Collembola	30	3.83 ± 0.13	3.14 ± 0.21	5.12 ± 0.38	4.05 ± 0.39	4.23 ± 0.42		30	3.26 ± 0.39	3.70 ± 0.47	3.24 ± 0.29	4.11 ± 0.62	
Hemiptera	30	0.64 ± 0.04^{AB}	0.96 ± 0.11^{A}	0.38 ± 0.13^{B}	0.29 ± 0.09^{BC}	0.61 ± 0.07^{BC}	**	40	0.65 ± 0.17	0.86 ± 0.19	0.50 ± 0.12	0.44 ± 0.12	
Coleoptera	40	0.62 ± 0.17	0.64 ± 0.21	0.38 ± 0.14	0.32 ± 0.08	0.73 ± 0.21		55	0.67 ± 0.16	0.73 ± 0.17	0.70 ± 0.18	0.99 ± 0.17	
Diptera	22	3.38 ± 0.06^{A}	3.39 ± 0.32^{A}	3.05 ± 0.40^{AB}	2.56 ± 0.10^{BC}	2.47 ± 0.14^{C}	**	27	2.21 ± 0.35	2.21 ± 0.24	2.58 ± 0.34	2.91 ± 0.17	
Abundance – foliage-active invertebrates													
All	28629	182.9 ± 41.46	453.00 ± 65.29	713.6 ± 25.1	574 ± 135.95	567.3 ± 51.79	*	17974	149.9 ± 32.01	214.7 ± 33.54	286.7 ± 56.05	334.6 ± 91	
Araneae	1889	18.44 ± 6.11	32.33 ± 1.90	49.3 ± 7.79	34.17 ± 8.07	31.33 ± 3.56		1631	18.14 ± 3.09	20.13 ± 2.23	25.48 ± 4.15	22.94 ± 5.26	
Acarina	5755	33.20 ± 13.68^{A}	62.50 ± 19.10^{AB}	176.60 ± 23.55^{C}	110.60 ± 23.96^{BC}	100.10 ± 12.97^{AB}	*	2614	7.19 ± 3.09^{A}	36.27 ± 6.77^{AB}	37.40 ± 11.34^{AB}	63.00 ± 22.82^{B}	*
Collembola	3789	20.22 ± 2.04	107.00 ± 32.69	56.30 ± 9.34	75.60 ± 14.21	73.20 ± 19.50		2718	33.80 ± 18.52	19.47 ± 7.39	19.20 ± 13.12	43.00 ± 30.90	
Orthoptera	125	1.56 ± 0.11	3.50 ± 0.50	1.89 ± 0.59	2.17 ± 1.10	2.27 ± 1.24		100	1.81 ± 0.44	0.87 ± 0.37	1.76 ± 0.45	0.67 ± 0.26	

Group	Lowland						Montane					
	n	mean age of regrowth (year)				*P*	n	mean age of regrowth (year)				*P*
		3	13	25	33	57		2	6	16	31	
Hemiptera	2846	20.67 ± 4.79^{A}	39.00 ± 2.32^{AB}	82.00 ± 8.66^{C}	39.78 ± 6.30^{AB}	64.80 ± 12.16^{BC}	*	2944	21.38 ± 7.52	45.07 ± 9.40	64.50 ± 25.25	25.83 ± 6.68	
Thysanoptera	2040	2.22 ± 0.52^{A}	50.00 ± 1.69^{BC}	64.10 ± 10.96^{B}	38.50 ± 4.12^{C}	30.00 ± 6.50^{C}	**	1378	8.81 ± 3.89	12.20 ± 1.78	38.30 ± 15.87	11.44 ± 3.23	
Coleoptera	1349	6.11 ± 1.58	21.67 ± 3.58	45.20 ± 12.66	16.39 ± 5.95	30.27 ± 8.67		426	3.29 ± 0.70	5.60 ± 1.07	7.00 ± 1.96	7.00 ± 3.56	
Diptera	9019	61.10 ± 16.63	77.83 ± 2.21	171.20 ± 13.08	206.60 ± 83.16	182.80 ± 42.98		5820	42.05 ± 9.24	54.90 ± 13.28	73.10 ± 18.82	143.30 ± 72.75	
Hymen.	1001	6.67 ± 1.32	17.00 ± 0.42	18.89 ± 2.03	20.78 ± 5.96	19.67 ± 5.42		557	5.76 ± 1.62	9.33 ± 1.47	8.38 ± 1.84	6.67 ± 1.33	
Formicidae	226	3.44 ± 0.95	7.67 ± 2.32	6.78 ± 1.26	3.39 ± 1.32	1.80 ± 1.04							
Taxon richness – foliage-active invertebrates													
All	372	34.56 ± 3.41^{A}	75.83 ± 5.17^{B}	77.00 ± 2.03^{B}	67.50 ± 4.98^{B}	71.00 ± 7.03^{B}	**	378	37.33 ± 4.92	52.27 ± 4.62	54.05 ± 6.20	49.72 ± 3.40	
Araneae	88	5.33 ± 0.76	10.67 ± 0.42	9.67 ± 0.45	9.83 ± 1.36	10.80 ± 1.31		82	8.86 ± 1.33	8.87 ± 0.20	9.29 ± 0.63	9.22 ± 1.25	
Acarina	33	2.89 ± 0.38^{A}	6.33 ± 0.63^{AB}	8.89 ± 0.44^{B}	7.33 ± 0.67^{B}	8.07 ± 1.16^{B}	*	38	2.48 ± 0.63^{A}	5.53 ± 0.72^{B}	4.95 ± 0.67^{B}	6.28 ± 0.99^{B}	**
Hemiptera	52	4.00 ± 0.58	6.83 ± 0.74	8.56 ± 0.38	6.67 ± 0.63	6.87 ± 1.20		81	4.86 ± 0.55	6.87 ± 0.65	7.38 ± 1.24	5.78 ± 0.49	
Coleoptera	53	2.44 ± 0.36^{A}	7.50 ± 0.74^{B}	5.44 ± 0.07^{BC}	4.61 ± 0.80^{AC}	5.60 ± 0.65^{BC}	*	46	2.14 ± 0.43	3.47 ± 0.65	3.76 ± 0.83	2.72 ± 0.42	
Diptera	30	8.00 ± 0.50	10.33 ± 0.85	10.22 ± 1.08	9.39 ± 0.95	9.27 ± 0.85		29	9.57 ± 0.91	9.73 ± 0.88	9.10 ± 0.91	9.89 ± 1.12	
Hymen.	24	4.11 ± 0.52	8.67 ± 0.42	8.00 ± 0.91	7.50 ± 0.76	7.53 ± 0.90		23	3.71 ± 0.88	4.47 ± 0.57	4.71 ± 0.58	4.22 ± 0.60	

Table 3.6 ANOSIM tests for differences in ground- and foliage-active invertebrate assemblages among regrowth age categories (Global *R*) and pairwise tests between regrowth age categories. Regrowth age categories at each moorland location are identified by their mean age (years) since last burn. [a]insufficient permutations to construct a 5% significance level test. **P* < 0.05, ***P* < 0.01, ****P* < 0.001.

Location	Test	Ground-active	Foliage-active
Lowland	Global *R*	0.42**	0.32*
	3 v 13 [a]	0.17	1.00
	3 v 25 [a]	1.00	1.00
	3 v 33	0.52*	1.00**
	3 v 57	0.94*	1.00*
	13 v 25 [a]	1.00	0.83
	13 v 33	0.17	-0.05
	13 v 57 [a]	1.00	0.20
	25 v 33	-0.25	-0.19
	25 v 57	0.37	0.04
	33 v 57	0.46**	0.07
Montane	Global *R*	0.22**	0.32***
	2 v 6	0.16	0.13
	2 v 16	0.39**	0.55**
	2 v 31	0.40**	0.41**
	6 v 16	0.14	0.32*
	6 v 31	0.12	0.07
	16 v 31	0.07	0.40**

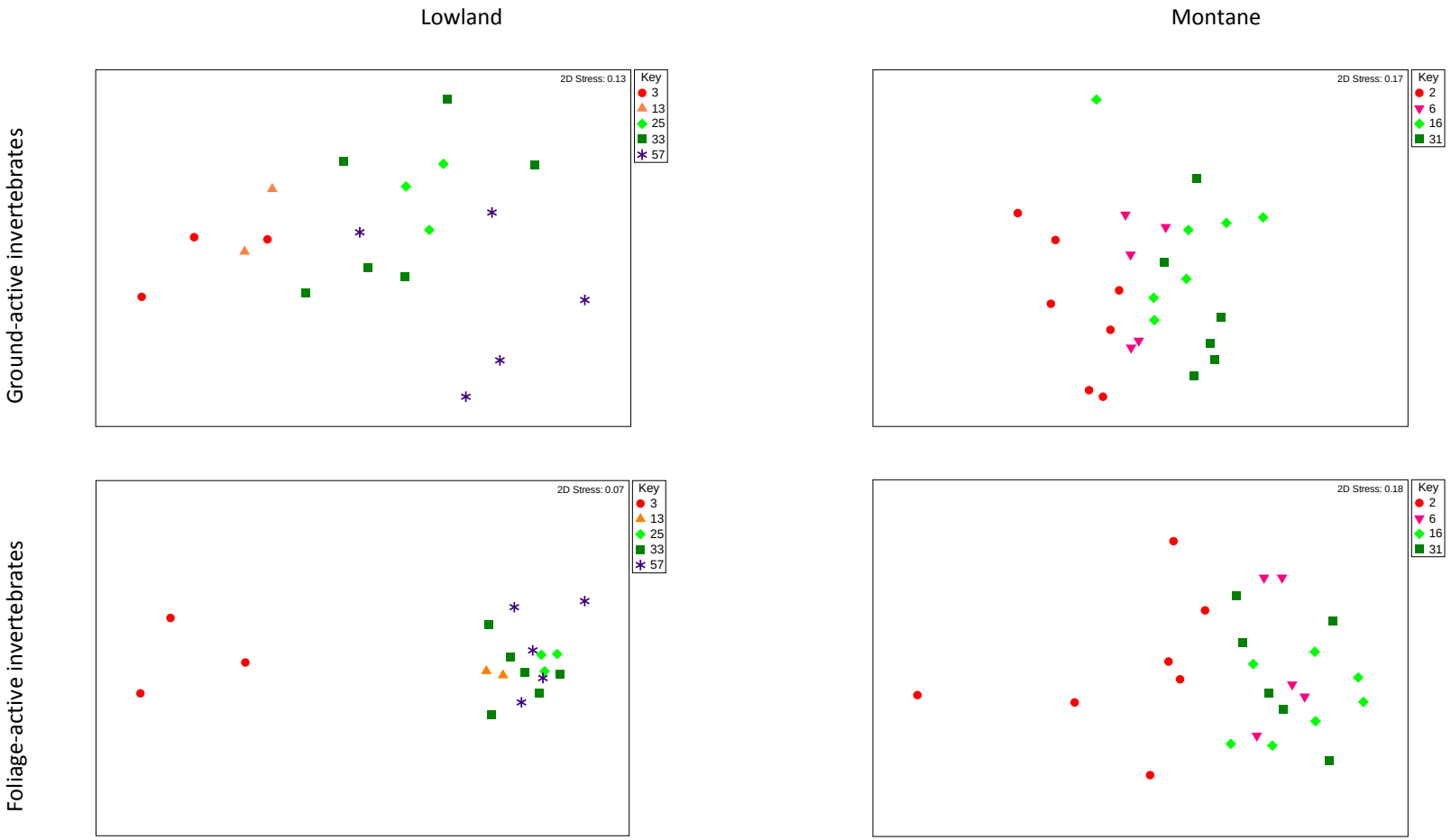

Fig. 3.2 MDS ordinations of sites at each moorland location based on log transformed ground- and foliage-active invertebrate abundances. Numbers shown in the keys are the mean ages (years) of the regrowth categories for sites**.** Note that different regrowth categories are used in lowland and montane moorlands**.**

Table 3.7 SIMPER analysis of individual taxon contributions to the difference in taxonomic composition between the youngest age of regrowth category and the combined older regrowth categories for ground-active invertebrates. Only taxa contributing more than 1% of mean dissimilarity are shown.

Lowland						Montane					
Group	Taxa	Mean age (year)		Contribution		Group	Taxa	Mean age (year)		Contribution	
		3	13–57	%	Cumm.%			2	6–31	%	Cumm.%
Collembola	*Corynephoria* sp. 1	11.30	0.79	6.42	6.42	Orthoptera	*Bobilla poene*	9.80	2.94	3.84	3.84
Orthoptera	*Bobilla poene*	9.49	1.61	4.63	11.05	Collembola	*Corynephoria* sp. 1	2.35	1.16	3.72	7.56
Collembola	*Sminthurinus* sp. 2	5.75	1.44	3.65	14.70	Collembola	Odontellidae spp.	7.85	2.86	3.51	11.07
Collembola	Odontellidae sp. 2	1.72	0.38	3.50	18.20	Collembola	*Acanthomurus* spp.	8.68	7.67	3.29	14.36
Collembola	Odontellidae spp.	6.17	3.48	3.48	21.68	Collembola	*Sminthurinus* sp. 2	0.86	0.28	2.43	16.79
Collembola	*Polykatianna* cf. *aurea*	2.86	1.20	2.98	24.66	Diptera	Cecidomyiidae spp.	1.75	0.60	2.11	18.90
Diptera	Cecidomyiidae spp.	4.05	1.25	2.71	27.37	Hymenoptera	*Iridomyrmex* sp. 1	1.29	0.38	1.99	20.89
Collembola	*Acanthomurus* spp.	38.65	20.12	2.20	29.57	Hymenoptera	Scelionidae spp.	0.58	1.20	1.81	22.71
Diptera	Muscidae larval spp.	1.16	0.26	1.99	31.55	Collembola	*Sminthurides* sp. 1	0.92	0.15	1.80	24.50
Diptera	Chloropidae spp.	1.18	0.20	1.98	33.53	Collembola	*Hypogastrura purpurescens*	0.23	0.55	1.74	26.24
Hymenoptera	*Anonychomyrma* ?*nitidiceps*	1.83	0.80	1.85	35.39	Diptera	Sphaeroceridae spp.	0.17	0.70	1.71	27.95
Diptera	Chironomidae spp.	1.01	0.19	1.75	37.14	Collembola	*Paronellides* sp. 5	0.06	0.73	1.62	29.57
Collembola	*Isotoma* sp. 1	0.05	0.67	1.56	38.70	Araneae	Lycosidae sp. 7	0.70	0.54	1.58	31.15
Collembola	Poduroida immatures	0.54	0.11	1.41	40.11	Collembola	Odontellidae sp. 1	0.57	0.22	1.54	32.69
Diptera	Sphaeroceridae spp.	0.02	0.55	1.40	41.52	Chilopoda	*Australeuma simile*	0.06	0.60	1.53	34.21
Diptera	Dolichopodidae spp.	0.70	0.21	1.39	42.91	Auchenorrhyncha	Euacanthellinae sp. 57	0.49	0.46	1.51	35.72
Heteroptera	*Systelloderis* sp. 61	0.49	0.01	1.30	44.21	Amphipoda	*Keratroides vulgaris*	0.38	0.88	1.50	37.23
Collembola	Collembolla indet.	0.84	0.35	1.21	45.42	Collembola	*Polykatianna* cf. *aurea*	0.72	0.55	1.45	38.68

Lowland						Montane					
Group	Taxa	Mean age (year)		Contribution		Group	Taxa	Mean age (year)		Contribution	
		3	13–57	%	Cumm.%			2	6–31	%	Cumm.%
Hymenoptera	Scelionidae spp.	1.01	0.93	1.17	46.59	Collembola	*Lepidophorella* sp. 1	0.17	0.57	1.41	40.09
Collembola	*Lepidocyrtus* sp. 1	0.39	0.73	1.09	47.68	Collembola	*Katianna* sp. 1	0.16	0.62	1.41	41.49
Collembola	*Sminthurinus* sp. 1	0.09	0.49	1.06	48.73	Diptera	Phoridae spp.	1.01	0.99	1.40	42.90
Amphipoda	*Neorchestia plicibrancha*	0.00	0.36	1.03	49.76	Trichoptera	*Tasmanoplegus spilota*	0.15	0.46	1.33	44.22
Prostigmata	*Halotydeus* sp. 1	0.02	0.36	1.01	50.77	Araneae	*Pardosa* sp. 4	0.54	0.23	1.24	45.47
Collembola	*Sminthurides* sp. 1	0.27	0.42	1.00	51.78	Diptera	Mycetophilidae spp.	0.19	0.43	1.15	46.62
						Oribatida	*Antarctozetes* nr. *longicaulis*	0.31	0.22	1.03	47.65

Table 3.8 SIMPER analysis of individual taxon contributions to the difference in taxonomic composition between the youngest age of regrowth category and the combined older regrowth categories for foliage-active invertebrates. Only taxa contributing more than 1% of mean dissimilarity are shown.

Lowland						Montane					
Group	Taxa	Mean age		Contribution		Group	Taxa	Mean age		Contribution	
		3	13–57	%	Cum.%			2	6–31	%	Cum.%
Thripidae	*Pseudanothrips achaetus*	0.67	26.66	2.83	2.83	Prostigmata	*Wartookia* sp. nov. 1	0.62	13.88	3.32	3.32
Prostigmata	*Wartookia* sp. nov. 1	0.00	15.28	2.76	5.60	Collembola	*Corynephoria* sp. 1	3.44	5.89	2.99	6.31
Auchenorrhyncha	Deltocephalinae sp. 29	9.49	0.02	2.40	8.00	Prostigmata	*Walzia australica*	0.52	9.49	2.79	9.10
Diptera	Ceratopogonidae spp.	9.59	99.48	2.34	10.34	Diptera	Empididae spp.	0.63	7.25	2.48	11.58
Collembola	*Rastriopes* sp. 1	0.49	12.33	2.29	12.63	Diptera	Rhagionidae spp.	5.36	0.65	2.21	13.79
Araneae	*Diaea rosea*	0.21	9.18	2.14	14.77	Heteroptera	Pachygronthidae sp. 1	3.31	8.68	2.07	15.86
Collembola	*Polykatianna* cf. *aurea*	3.71	35.97	2.09	16.86	Thripidae	*Pseudanothrips achaetus*	1.61	6.39	2.01	17.87
Auchenorrhyncha	Ulopinae sp. 18	0.00	7.08	2.09	18.95	Diptera	Ceratopogonidae spp.	2.82	7.08	1.87	19.73
Thripidae	*Thrips imaginis*	0.26	7.58	1.90	20.86	Araneae	*Dictyna* sp. 1	1.46	2.67	1.57	21.30
Oribatida	*Baloghobates* sp. 1	2.16	12.74	1.84	22.70	Diptera	Chironomidae spp.	11.68	20.33	1.55	22.85
Heteroptera	Acanthosomatidae sp. 6	0.00	5.17	1.82	24.52	Sternorrhyncha	Sternorrhyncha indet.	0.67	2.42	1.50	24.34
Prostigmata	*Walzia australica*	0.21	6.46	1.79	26.31	Thripidae	*Thrips imaginis*	2.86	4.70	1.47	25.82
Coleoptera	*Pseudomicrocaria* sp. 1	0.31	6.54	1.73	28.03	Heteroptera	Acanthosomatidae sp. 6	0.04	1.80	1.39	27.21
Heteroptera	Acanthosomatidae sp. 104	0.00	3.81	1.58	29.61	Diptera	Chamaemyiidae spp.	2.82	0.73	1.36	28.57
Psocoptera	Psocoptera spp.	4.00	0.23	1.52	31.13	Collembola	*Polykatianna* cf. *aurea*	3.48	2.67	1.36	29.93
Araneae	*Dictyna* sp. 1	0.00	3.62	1.48	32.61		Wartookia sp. 2	0.08	1.75	1.30	31.23
Coleoptera	*Macrohelodes* sp. 3	0.11	3.71	1.47	34.08	Diptera	Tipulidae spp.	2.25	2.06	1.11	32.33
Diptera	Muscidae spp.	5.55	0.60	1.45	35.53	Coleoptera	*Pseudomicrocaria* sp. 1	0.99	1.36	1.05	33.38

Lowland						Montane					
Group	Taxa	Mean age		Contribution		Group	Taxa	Mean age		Contribution	
		3	13–57	%	Cum.%			2	6–31	%	Cum.%
Diptera	Chironomidae spp.	24.79	5.36	1.41	36.94	Araneae	*Eriophora pustulosa*	1.12	2.06	1.04	34.42
Diptera	Empididae spp.	1.36	7.85	1.36	38.30	Diptera	Tephritidae spp.	1.41	1.94	1.02	35.44
Diptera	Rhagionidae spp.	0.00	2.86	1.32	39.62	Araneae	*Tetragnatha valida*	2.29	2.35	1.00	36.44
Coleoptera	*Monolepta* sp. 1	0.00	2.53	1.28	40.90	Araneae	*Araneus arenaceus*	0.77	1.25	1.00	37.44
Collembola	*Corynephoria* sp. 1	15.44	5.36	1.26	42.16						
Hemiptera	Hemiptera nymphs	0.52	4.26	1.25	43.40						
Oribatida	*Baloghobates* sp. 2	11.68	33.12	1.24	44.64						
Hymenoptera	Encyrtidae spp.	0.00	2.49	1.23	45.88						
Oribatida	Oribatida nymphs	5.89	16.29	1.16	47.04						
Heteroptera	Pachygronthidae sp. 1	1.56	5.49	1.15	48.19						
Hymenoptera	Eulophidae spp.	1.12	4.47	1.04	49.24						
Araneae	*Eriophora* sp. 2	2.03	0.28	1.01	50.25						
Sternorrhyncha	Sternorrhyncha indet.	0.19	2.00	1.01	51.26						

Table 3.9 ANOSIM tests for differences in vegetation composition among regrowth age catgories (Global *R*) and pairwise tests between regrowth age catgories for each moorland location. Regrowth age catgories at each location are identified by their mean age (years) since last burn. [a]insufficient permutations to construct a 5% significance level test. **P* < 0.05, ***P* < 0.01, ****P* < 0.001.

Location	Test	*R*
Lowland	Global *R*	0.51***
	3 v 13 [a]	0.83
	3 v 25 [a]	1.00
	3 v 33	1.00**
	3 v 57	0.93*
	13 v 25 [a]	1.00
	13 v 33	0.81*
	13 v 57 [a]	0.46
	25 v 33	0.11
	25 v 57	-0.22
	33 v 57	0.07
Montane	Global *R*	0.36***
	2 v 6	0.19
	2 v 16	0.51**
	2 v 31	0.44*
	6 v 16	0.49**
	6 v 31	0.70**
	16 v 31	0.13

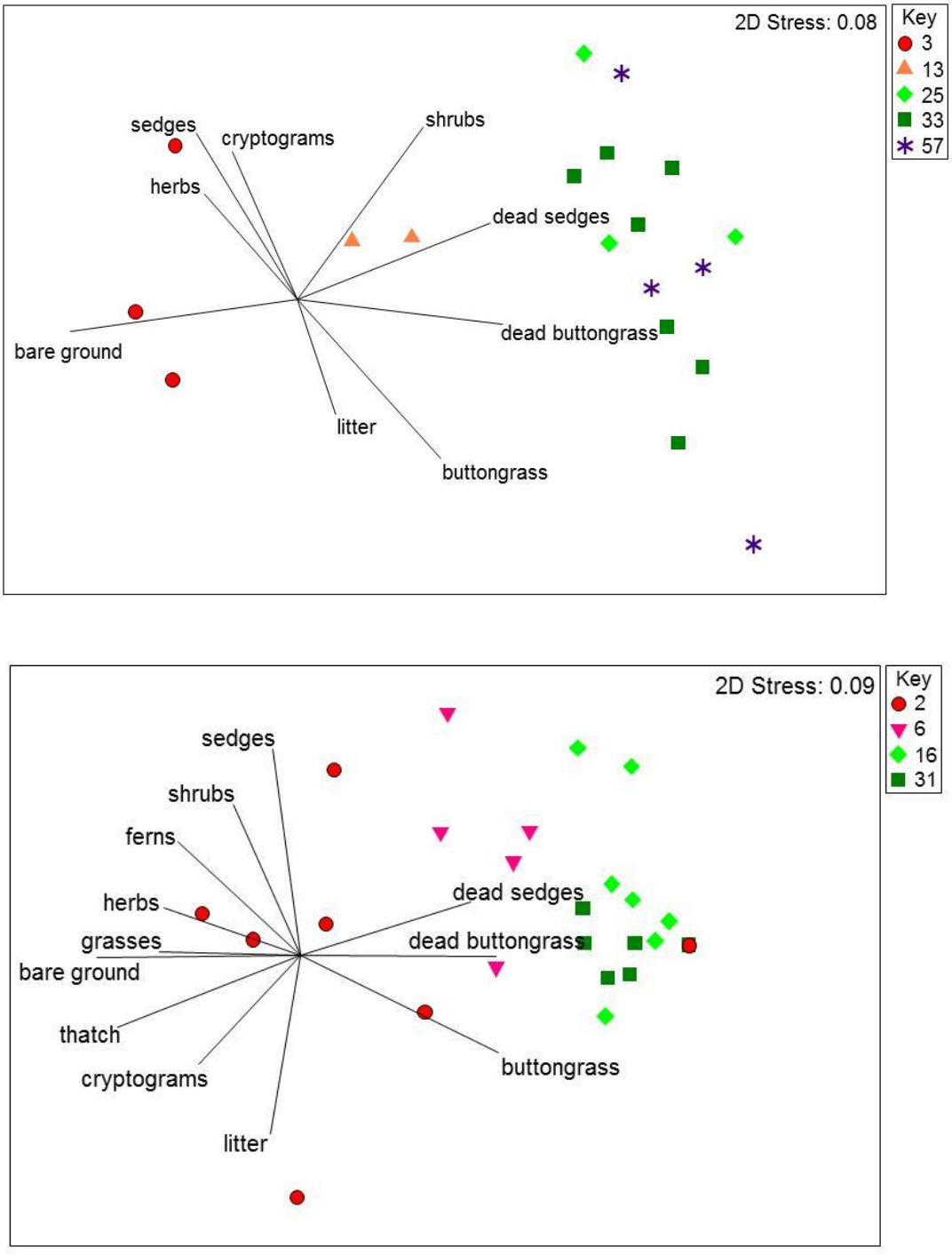

Fig.3.3 MDS ordination of sites in lowland moorland (top) and montane moorland (bottom) based on square root transformed cover scores for vegetation variables. Numbers shown in the keys are the mean ages (years) of the regrowth categories for sites**.** Vectors show the direction and strength of Pearson correlations with vegetation variables (only correlations ≥ 0.5 are shown).

Table 3.10 Mean (±se) values for environmental variables for each age of regrowth category at each moorland location. Only variables that differed significantly between age of regrowth using one-way ANOVAs are shown (**P* < 0.05, ***P* < 0.01, ****P* < 0.001). Categories that do not share the same letters are significantly different from each other based on post-hoc pairwise comparisons using Fisher's least significant difference method.

Lowland							Montane					
Variable	Age of regrowth (years0					*P*	Variable	Age of regrowth (years)				*P*
	3	13	25	33	57			2	6	16	31	
Vegetation density	17.14 ± 1.89^{A}	41.07 ± 3.22^{AB}	95.00 ± 8.22^{BC}	87.65 ± 9.66^{BC}	129.80 ± 30.65^{C}	**	Vegetation density	33.88 ± 6.80^{A}	70.43 ± 3.85^{B}	72.24 ± 5.72^{B}	95.00 ± 3.50^{C}	***
Total nitrogen (%)	0.76 ± 0.17^{ABC}	1.13 ± 0.28^{A}	0.47 ± 0.13^{C}	0.82 ± 0.08^{AB}	0.91 ± 0.07^{BC}	*	Total nitrogen (%)	1.26 ± 0.25^{AB}	1.22 ± 0.13^{AB}	1.49 ± 0.20^{A}	0.97 ± 0.42^{B}	*
Calcium (ppm)	423.3 ± 125.0^{AB}	565.0 ± 77.8^{A}	220.0 ± 20.0^{AB}	394.3 ± 154.1^{B}	240.0 ± 107.4^{B}	*	Conductivity (µs/cm)	61.29 ± 12.79^{AB}	61.20 ± 8.44^{AB}	76.43 ± 14.62^{A}	48.80 ± 27.30^{B}	*
Peat depth (cm)	27.86 ± 6.24^{A}	28.57 ± 3.84^{AB}	40.86 ± 5.70^{AB}	38.08 ± 5.13^{A}	50.89 ± 15.77^{B}	*						

Table 3.11 The contribution (R^2) of each environmental predictor variable to explaining the variation in invertebrate assemblages and their relative importance when their AIC_c weights ($\sum\omega_i$) are summed across all DISTLM models in the multi-model approach. Highest ranked variables for each assemblage are in bold. Global model is the contribution of all eight variables to explaining the variation in invertebrate assemblages. *P < 0.05 from marginal test (variables fitted alone).

	Lowland moorland				Montane moorland			
	Ground-active		Foliage-active		Ground-active		Foliage-active	
	R^2	$\sum\omega_i$	R^2	$\sum\omega_i$	R^2	$\sum\omega_i$	R^2	$\sum\omega_i$
Peat depth	0.13*	0.31	0.11	0.26	0.09*	0.36	0.06	0.31
Vegetation density	0.22*	**0.53**	0.27*	**0.74**	0.11*	**0.60**	0.11*	**0.58**
Total area	0.06	0.29	0.05	0.31	0.09*	0.40	0.05	0.33
Closest forest	0.05	0.27	0.04	0.26	0.04	0.29	0.04	0.29
Number of fires	0.12*	0.31	0.09	0.26	0.11*	0.49	0.08*	0.40
Soil PC1	0.05	0.22	0.04	0.25	0.10*	0.37	0.08*	0.42
Soil PC2	0.18*	0.42	0.17*	0.34	0.05	0.34	0.04	0.34
Veg. PC2	0.07	0.31	0.6	0.29	0.06	0.29	0.06	0.40
Best model[1]								
Vegetation density	0.22		0.27		-		-	
Vegetation density + number of fires	-		-		0.23		0.20	
Global Model	0.58		0.57		0.50		0.46	

[1]lowest AIC_c ranked model

Table 3.12 The best three matches between the among-sample patterns for each of four invertebrate assemblages and that from environmental variables associated with those samples using BIOENV. The match with vegetation density (VD) alone is also shown. *ρ*, spearman correlation coefficient; VD, vegetation density; PD, peat depth; S1, Soil PC1; S2, Soil PC2; NF, number of fires; TA, total area; V1, Veg. PC1; V2, Veg. PC2. Statistical significance of top ranked matches is 0.001 except for montane foliage-active assemblage which was 0.01 (4,999 permutations).

Lowland				Montane			
Ground-active		Foliage-active		Ground-active		Foliage-active	
Best match	*ρ*	Best match	*ρ*	Best match	*ρ*	Best match	*ρ*
VD + PD +S2	0.64	VD	0.67	VD + TA + NF + S1	0.59	VD + S1 + V2	0.43
VD + PD + S2 + NF	0.62	VD + S2	0.59	VD + TA + NF	0.58	VD + V2	0.43
VD + PD	0.62	VD + PD	0.55	VD + TA + NF + PD	0.58	VD + V1	0.43
VD	0.56		0.67	VD	0.39	VD	0.42

Plate 14 Vegetation density three years after fire in lowland moorland

Plate 15 Vegetation density 54 years after fire in lowland moorland

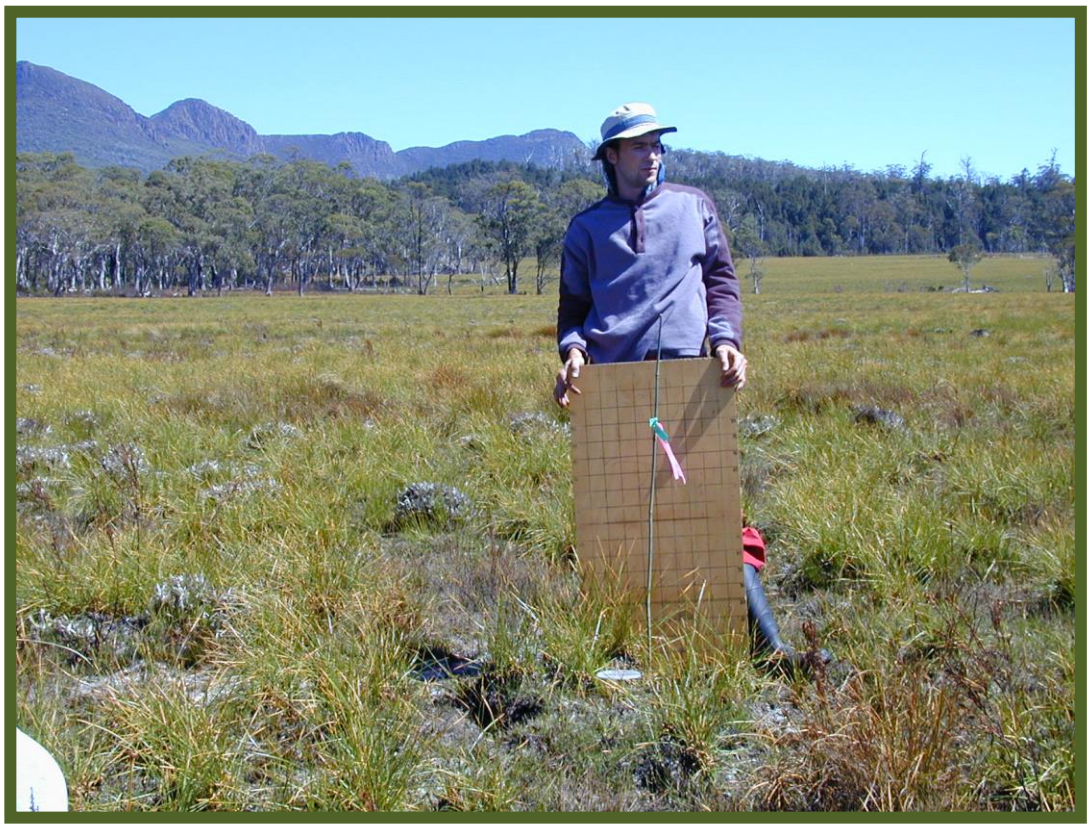

Plate 16 Vegetation density one year after fire in montane moorland

Plate 17 Vegetation density 31 years after fire in montane moorland

Compositional changes in invertebrate assemblages

There was an age of regrowth effect on assemblage composition for both ground- and foliage-active invertebrates in both lowland and montane moorlands (Table 3.6, Fig. 3.2). In all cases, assemblages in the youngest age of regrowth (≤ 3 years) were clearly distinguished from assemblages in older regrowth, especially in lowland moorland. Invertebrate assemblages in older regrowth overlapped strongly; however, for ground-active invertebrates in lowland moorland there was evidence that assemblages in the oldest regrowth (57 years) were different from the assemblages in most of the younger regrowth sites. In general there was greater overlap of invertebrate assemblages in montane moorland than in lowland moorland (Table 3.6, Fig. 3.2).

The taxa making the most contribution to the difference in taxonomic composition between the youngest regrowth categories and the older regrowth categories were similar for lowland and montane moorlands for ground- and foliage-active assemblages (Tables 3.7–3.8). Excluding rarely caught taxa, no taxa were completely absent from the combined older regrowth categories and very few were completely absent from the youngest regrowth category (Tables 3.7–3.8). Among ground-active invertebrates, only *Neorchestia plicibrancha* (Amphipoda) was absent in the youngest regrowth category (Table 3.7). Eight foliage-active taxa were absent in the youngest regrowth category and these were all in lowland moorland: *Wartookia* sp. nov. 1 (Acarina), *Dictyna* sp. 1 (Araneae), Ulopinae sp. 18 (Auchenorrhyncha), Acanthosomatidae sp. 6 and sp. 104 (Heteroptera), *Monolepta* sp. 1 (Coleoptera), Rhagionidae spp. (Diptera) and Encyrtidae spp. (Hymenoptera) (Table 3.8).

Changes in environmental variables with age of regrowth

In both lowland and montane moorland there was a clear shift in vegetation composition with increasing age of regrowth, with the separation of regrowth sites much greater in lowland moorland than in montane moorland (Table 3.9, Fig. 3.3). In lowland moorland, the 3 year old regrowth sites were strongly separated (R = 0.83–1.00) from all older regrowth sites. The 13 year old regrowth sites were strongly separated from the 25 and 33 year old regrowth sites, and moderately

separated from the 57 year old regrowth site. The vegetation at the older lowland regrowth sites (25, 33 and 57) was compositionally indistinguishable. In montane moorland the 2 and 6 year old regrowth sites were barely distinguishable but both were separated from older regrowth sites. In both lowland and montane moorlands the youngest regrowth sites had more bare ground and more cover of ferns, herbs, cryptograms and grasses than older regrowth sites (Fig. 3.3). Older regrowth sites were dominated by buttongrass and dead buttongrass and other sedges and, in lowland moorland only, by shrubs (Fig. 3.3).

There was no significant effect of age of regrowth on most environmental variables ($P > 0.05$). However, age of regrowth had a strong, significant effect on vegetation density and this effect was similar for both locations with less vegetation density in the youngest regrowth sites than in the older regrowth sites (Table 3.10, Plates 14–17). Age of regrowth also had a significant effect on total nitrogen, calcium, conductivity and peat depth (Table 3.10) but the effects differed between locations and, with the possible exception of peat depth, are not readily interpreted suggesting confounding with other unknown factors. At the lowland location peat depth was shallower at the youngest sites than at the oldest sites.

Invertebrate assemblage structure was related to the among-sample vegetation structure at both locations, with the correlation stronger for foliage-active (lowland: $\rho = 0.57$, $P < 0.001$; montane: $\rho = 0.50$, $P < 0.001$) than ground-active invertebrate assemblages (lowland: $\rho = 0.43$, $P < 0.001$; montane: $\rho = 0.29$, $P < 0.01$).

Environmental predictors of invertebrate assemblage composition

For both locations and for both ground- and foliage-active invertebrates, large proportions (46–58%) of the variation in invertebrate assemblage composition could be explained by the eight environmental predictor variables (Table 3.11). The 'best' model in lowland moorland for both ground- and foliage-active assemblages was vegetation density, whereas the 'best' model in montane moorland for the two invertebrate assemblages was vegetation density and number of fires. However, there were a large number of models within seven units of $AIC_{c(min)}$. Summed AIC_c weights indicated that vegetation density was the best predictor variable for

ground- and foliage-active invertebrates at both locations. Vegetation density was highly correlated with Veg. PC1 (lowland and montane), age of regrowth, total vegetation cover and vegetation height. There was either little support or considerable uncertainty for the other predictor variables, with the possible exception that number of fires may be an important predictor of variation for montane, ground-active invertebrate assemblages. All eight predictor variables (global model) and vegetation density ('best' predictor variable) explained larger proportions of variation for invertebrate assemblages in lowland moorland than in montane moorland.

The BIOENV routine also identified vegetation density as the best model for the foliage-active invertebrate assemblages at both locations (Table 3.12; note, on the grounds of parsimony ρ = 0.42 for vegetation density is a better match than ρ = 0.43 for vegetation density + Soil PC1 + Veg. PC2). The BIOENV routine also identified peat depth as an important predictor variable for lowland, ground-active invertebrates and number of fires + total area of buttongrass for montane, ground-active invertebrates.

3.5 Discussion

Changes in invertebrate composition, abundance and taxon richness with age of regrowth

As expected (Moretti et al. 2002; Moretti et al. 2004; Brennan et al. 2006; Pyrke and Samways 2012a; Kim and Jung 2013), fire altered the composition of ground and foliage-active invertebrate assemblages in both lowland and montane moorlands, with assemblages in the youngest regrowth categories clearly distinguished from those assemblages in older regrowth categories. Contrary to expectations (Springett 1976; Abbott 1984; Sieman et al. 1997; Bess et al. 2002; Moretti et al. 2004; Coleman and Rieske 2006; Barratt et al. 2009), total abundance of ground-active invertebrates was highest in the youngest regrowth categories, although foliage-active invertebrates followed the expected trend (i.e. lowest abundance in youngest regrowth). A similar increase in total abundance of ground-active invertebrates (based on 13 invertebrate orders) was found one year after fire in South African fynbos and this was attributed to a large increase of Formicidae (Pyrke and

Samways 2012b, a); however, unlike the present study, the Orthoptera did not increase in abundance and Collembola and Diptera were not surveyed. It is possible that the present study missed an initial short-term (≤1 year) decrease in taxon richness and abundance following fire that has been reported in studies in a range of habitats (Brennan et al. 2006; Barratt et al. 2009; Underwood and Quinn 2010; Pyrke and Samways 2012a). Comparison of trends in total invertebrate abundance and taxon richness found in the present study with the few previous multiorder studies is problematic because invertebrate orders, sampling methods, fire history and vegetation communities vary.

The effects of fire on buttongrass moorland invertebrates found in the present study are broadly consistent with most previous studies in buttongrass moorland (Greenslade and Driessen 1999; Driessen and Greenslade 2004; Driessen et al. 2013). However, in a study of Acarina sampled using soil cores, using the same survey sites that were used in the present study, Green (2008, 2009) found that Acarina density and richness increased 30+ years post-fire, suggesting that soil-active Acarina may take a long time to re-establish after fire. The present study did not detect an increase in Acarina abundance 30+ years post-fire. However, only three of the five most dominant species collected from soil cores were collected in pitfall samples and these were represented by less than five specimens; indicating that the soil-active Acarina assemblage is markedly different to the ground- and foliage-active Acarina community. A complete description of the soil-active fauna has yet to be published and further investigation of the response to fire by soil-active invertebrates is required. The different responses to fire by soil, ground- and foliage-active Acarina highlight the importance of using different methods to survey a range of invertebrate groups and also to survey a range of species within an invertebrate group.

Pitfall traps and sweep nets sampled different components of the moorland invertebrate community to the extent that different responses to fire occurred within the same order and family depending on survey method. The need to use a range of methods and invertebrate taxa (or functional groups) when conducting investigations into the effects of fire, or other disturbance, on invertebrates is well-recognised (Pyrke and Samways 2012a, b; Teasdale et al. 2013).

Environmental variables potentially important in predicting changes in invertebrate assemblages

Determining how the vegetation variables associated with faunal variation directly influence invertebrate assemblage composition in buttongrass moorlands will require manipulative experiments. However, plant community composition, plant species diversity and plant structural diversity are all well-recognised predictors of variation in invertebrate abundance and species richness in successional processes (Hutchinson 1959; Murdoch et al. 1972; Southwood et al. 1979; Lawton 1983; Siemann et al. 1998; Knops et al. 1999; Walter and Proctor 1999; Koricheva et al. 2000; Haddad et al. 2001; Richardson and Hanks 2009; Price et al. 2011). Changes in vegetation cover, composition and architecture can affect life-cycle processes and survival of invertebrates through changes in microclimate conditions, such as temperature, humidity and illumination, and through the availability of food (including feeding niches) and shelter (Lawton 1983; Evans 1984; Majer 1984; Warren et al. 1987; Hulbert 1988; Niwa and Peck 2002; Hochkirch and Adorf 2007; Price et al. 2011).

The number of previous fires has been recognised as an under-researched predictor for biotic responses to fire management (Gill et al. 2002). The lack of any predictive power for this variable at the lowland site relates to a low degree of variation in the number of previous fires across my study sites (Table 3.2). At the more variable montane moorland location the number of previous fires predicted variation in invertebrate assemblage structure, particularly for ground-active invertebrates, a result consistent with previous work (Moretti et al. 2002; Hanula and Wade 2003; Wallner et al. 2012), although in tallgrass prairies fire frequency had minimal effects on ground beetle diversity measures (Cook and Holt 2006).

Peat depth may have potential in predicting variation in lowland ground-active assemblages. It may have direct effect by providing increased volume and or quality of habitat for ground-active invertebrates, particularly soil-active invertebrates. Soil depth, in particular, the depth of organic material, is known to influence invertebrate assemblage composition (Mitchell 1978; Giller 1996; Marra and Edmonds 1998; Maraun and Scheu 2000). Peat depth may also be a proxy for

other variables that could influence invertebrate assemblages in buttongrass moorland, such as drainage and topography (Nicolls and Dimmock 1965; di Folco 2007), although topographic differences were minimal in the present study.

The availability of litter is an important factor predicting ground-active invertebrate assemblage structure (York 2000; Parr et al. 2004; Brennan et al. 2006), but was probably too low to have an effect in buttongrass moorlands. Similarly to our moorlands, the cover of litter in European peat bogs was not affected by fire (Hochkirch and Adorf 2007). In the present study dead and decaying sedges filled the role of litter, increasing with time and being associated with changes in taxon composition.

Soils can have an influence on the composition of ground- and foliage-active invertebrate assemblages either directly, or indirectly through its influence on vegetation structure and composition (Ruston et al. 1991; Stork and Eggleton 1992; McCraken 1994; Sanderson et al. 1995; Giller 1996), and through its effect on food quality of plants (Prestidge 1982; Sedlacek et al. 1988; Stork and Eggleton 1992; Sanderson et al. 1995). The limited influence of the soil variables on variation in invertebrate assemblage structure within each of the lowland and montane locations probably relates to my sampling strategy, which sought environmental uniformity between sites. However, the differences in soil variables between the lowland and montane locations may have had a major influence on the rate of return to pre-fire states for both ground- and foliage-active invertebrates, as it does for vegetation (Bowman et al. 1986, present paper). The difference in invertebrate assemblage return time to the pre-fire states between the two moorland types provides explicit evidence that it is not time *per se* that is important for the return of invertebrates to the pre-fire state but the return of vegetation and related environmental variables to the pre-fire state.

Limitations

Important caveats on these findings are: (1) many invertebrate taxa were not surveyed at all or were not effectively surveyed by these methods (e.g. soil-active and winter-active species) and their responses to fire are largely unknown, (2) the moorland communities have been influenced by a long history of fire that date back

many thousands of years and fire sensitive taxa may already have been lost, and (3) there are many aspects to a fire regime that have only been partly (e.g. fire-frequency) or not (e.g. fire intensity) addressed by this study.

3.6 Conclusions

In one of the very few investigations of fire effects on invertebrates that have used a broad range of groups (34 orders) and a wide range of fire ages, I found that vegetation density and several other environmental variables were better predictors of assemblage response to fire than time elapsed since fire *per se*. Overall fire in buttongrass moorland had a limited impact on ground-active and foliage-active invertebrates, suggesting that these assemblages are resilient to single fires. Although several common foliage active taxa were absent in moorland for several years after fire they re-colonised sites as the vegetation returned to the pre-fire state, indicating that individual fires are unlikely to constitute a threat to the invertebrate biota.

Plate 18 Badge huntsman spider *Neosparassus* sp. (Araneae: Sparassidae)

Chapter 4

Higher taxa are effective surrogates for species-level data in representing patterns of invertebrate assemblage structure due to disturbance in moorlands.

4.1 Abstract

Higher taxa have been widely used in biodiversity studies as a cost-effective surrogate for species-level identification. Few studies have compared their effectiveness with species-level data for terrestrial invertebrates, and virtually all of these studies have focused on one or two orders. This study used a terrestrial invertebrate dataset comprising 21 orders, 197 families and 752 species to investigate whether order and family level identifications were effective surrogates for species-level identification in representing patterns in assemblage structure and detecting the effects of fire. Factors potentially influencing the effectiveness of surrogates among invertebrate orders were also investigated. Family-level identification of invertebrates in moorland sites with a wide range of fire history was found to be an effective surrogate for species-level identification. Order-level identification was also an effective surrogate but the level of discrimination among sites was typically lower than for species- or family-level identification and it may not detect more subtle changes as a result of disturbance. Higher taxonomic surrogates performed well for invertebrate orders comprising a few species that were abundant and with a small mean and variance in the number of species per higher taxa. Higher taxonomic surrogates are a useful, cost-effective approach for monitoring impacts of disturbance but their successful application will depend on taxonomic diversity, community structure and, ultimately, the objectives of the study.

Key words: higher taxon approach, fire, taxonomic sufficiency, biodiversity assessment

4.2 Introduction

The high cost of species-level identification for biodiversity assessment and environmental disturbance studies has led to the investigation of surrogates such as indicator taxa, environmental indicators and higher taxa (Noss 1990; Gaston and Williams 1993; Balmford et al. 1996; Basset et al. 2004). This is particularly the case for invertebrate studies which are also often constrained by insufficient taxonomic knowledge and the limited availability of taxonomists. One of the more widely used and arguably more successful surrogates is higher taxa; that is using genus or higher taxonomic level identification in place of species. In a review of almost 300 case studies, covering a range of environments and organisms, Bevilacqua et al. (2012) found that higher taxa up to at least family-level can be successful in identifying patterns of community change due to natural or human disturbance. The performance of higher taxa as surrogates for species will, in large part, depend on the degree of environmental heterogeneity being considered, with higher taxonomic levels more likely to represent species-level patterns if perturbations are large or environmental gradients are distinct (Olsgard et al. 1998; Biaggini et al. 2007; Nakamura et al. 2007; Schipper et al. 2010; Bevilacqua et al. 2012). The performance of higher taxa is also strongly influenced by the ratio of the number of higher taxa to the number of species (Bevilacqua et al. 2012; van Rijn et al. 2015). Bevilacqua et al. (2012) found that the effectiveness of higher taxonomic surrogates was very low when the ratio of the number of higher taxa to the number of species (ø) was less than 0.4. Similarly, for bee data sets across three biomes and various habitats, van Rijn et al. (2015) found that the number of species per higher taxon was a main factor influencing higher taxa performance. Using a modelling approach, Neeson et al. (2013) found that higher taxonomic surrogates are also negatively affected by high variance in the distribution of individuals among species with higher taxa surrogates performing best in communities in which a few common species are most abundant. Ultimately, the successful application of higher taxonomic surrogates will also depend on the objectives of the study (Ellis 1985; Lenat and Resh 2001; Schipper et al. 2010).

Higher taxonomic surrogates have been most frequently applied in marine and freshwater environments particularly in relation to studies of human impacts

on invertebrates (Lovell et al. 2007; Bevilacqua et al. 2012). Although higher taxa have been used to investigate effects of disturbance on invertebrate assemblage structure in terrestrial environments, very few studies have assessed their effectiveness in relation to patterns based on species-level data. Of those that have, virtually all have focussed on only one or two orders (Brennan et al. 2006; Mandelik et al. 2007; Rosser and Eggleton 2012; Vieira et al. 2012; Timms et al. 2013) or have compared assemblage patterns based on diverse order-level data with patterns for species-, genera- and family-level data within a single order (Biaggini et al. 2007; Schipper et al. 2010). Using one or two orders is of limited value if the purpose is to investigate biodiversity responses to disturbance; especially as studies have shown that invertebrate groups vary in their response to disturbance (Pyrke and Samways 2012b). Thus, there is a need for studies assessing the effectiveness of higher taxa surrogates using a broad range of invertebrate taxa. In South African savannas Lovell et al. (2007), using nine invertebrate groups, found that higher taxa (family, genus) were good surrogates for species-level identification in diversity and assemblage patterns.

The present study investigates whether higher taxonomic levels (abundance data aggregated to order and family level) are effective in representing multivariate patterns in invertebrate assemblage structure based on species abundance data amongst moorlands subject to different fire regimes. Because fire is a major disturbance event, I expect that higher taxa will be effective surrogates for species-level identification and that data aggregated to family level will be more effective than data aggregated to order level. I also tested the hypothesis that the number of species per higher taxa and the number of individuals per species were important predictors of the effectiveness of higher taxonomic surrogates. Other potential predictors relating to taxonomic diversity and community structure were also tested

4.3 Methods

Data were used from a replicated space-for-time substitution study that investigated the effect of fire on moorland invertebrates (Chapter 3). The study was conducted in Tasmania, Australia in lowland (320 m) buttongrass moorland (19

sites) and in montane (800 m) buttongrass moorland (25 sites). The sites have different time since fire history and were assigned *a priori* to age of regrowth categories (Table 4.1). In late summer, ground-active and foliage-active invertebrate abundance data were collected using pitfall traps (seven per site) and sweep nets (three by 100 sweeps per site) respectively. Invertebrates were identified by taxonomists (listed in Appendix 1) to species or morphospecies (i.e. morphologically distinguishable). For convenience, 'species' will be used in this chapter to refer to both species and morphospecies. Using a dataset comprising over 800 species, I detected a clear response to fire, with invertebrate assemblage structure in the youngest regrowth clearly distinguished from older regrowth for both ground- and foliage-active invertebrates in both lowland and montane moorland (Chapter 3).

Table 4.1 Age range, mean age (years) and number of sites (n) for each regrowth category for lowland and montane moorland.

Category	Lowland			Montane		
	Age range	Mean age	n	Age range	Mean age	n
1	3–3	3	3	1–3	2	7
2	-	-	-	5–8	6	5
3	12–13	13	2	13–20	16	7
4	22–27	25	3	-	-	-
5	31–40	33	7	30–31	31	6
6	54–65	57	4	-	-	-

To investigate higher taxonomic surrogates, groups of invertebrates that were not identified to species in Chapter 3 were excluded from the analysis (Diptera, non-ant Hymenoptera, Psocoptera, Neuroptera, Ephemeroptera, Oligochaeta). Immature Collembola, Hemiptera and Thysanoptera usually could not be identified below family level and were also excluded from the analysis. Larvae of Lepidoptera and Coleoptera were identified and counted separately from adults even where adults and larvae of the same species were identified. For taxa with similar-looking immatures and adults (e.g. arachnids, paurometabolous insects) these life stages were counted together. Invertebrate species abundance data were

aggregated to the levels of family and order. Full details of study sites and sampling methods are given in Chapter 3.

All multivariate analyses were carried out using the PRIMER version 6.1 computer program (Clarke and Gorley 2006). Assemblages from the two locations and sampling methods were analysed separately because of large differences among these assemblages (Chapters 2–3). Invertebrate abundance data were log (x + 1) transformed to down-weight the contributions of quantitatively dominant species and used to generate Bray–Curtis similarity measures. To visualise multivariate patterns in invertebrate assemblage structure, multidimensional scaling (MDS) was performed. The RELATE routine was used to test if the among-sample relationships for species abundance data were similar to the among-sample relationships for abundance data aggregated to family and order level. Following Lovell et al. (2007), if $\rho > 0.75$ then higher taxonomic surrogates were considered effective. One-way analyses of similarity (ANOSIM) (Clarke and Warwick 2001) using 4,999 permutations were performed on the data to test for differences in the invertebrate assemblage composition between age of regrowth categories. ANOSIM returns an *R*-statistic which gives a measure of how similar categories are. R-values most commonly range from 0 to 1; the closer the *R*-value is to 1 the more different the categories are, while a value close to zero indicates assemblages can barely be separated (Clarke and Warwick 2001). Large pairwise *R* values were used to indicate major differences between categories provided the global test was significant (Clarke and Gorley 2006). ANOSIM tests are robust to differences in the number of samples in factor groups (Clarke and Warwick 2001). Abundance data aggregated to family and order level were considered effective if the analyses to detect significant ($P < 0.05$) effects of age of regrowth on assemblage structure were consistent with analyses based on species abundance data.

Predictors of the effectiveness of higher taxonomic surrogates

The distance-based linear model (DISTLM) routine (Anderson et al 2008) was used to explore and model the relationship between the effectiveness of higher taxonomic surrogates (order and family) and several predictors relating to taxonomic diversity (number of species per higher taxa, and its standard deviation

and coefficient of variation) and community structure (number individuals per species and its standard deviation and coefficient of variation, and the number of individuals of the most common species as a percentage of the total number of individuals (commonest species)). Because there was only one order per sample, standard deviation for number of species per order could not be determined. Effectiveness is defined as the correlation (spearman rank correlation, ρ) between species and higher-taxon matrices. Only taxonomic orders with ≥25 individuals were used in the analyses. Values for predictor variables are given in Appendix 3.

Draftsman's plots were used to examine the distribution of variables and to identify highly correlated variables ([r]) ≥ 0.8). The number of species per higher taxa and the number of individuals per species were fourth root transformed prior to analyses. Commonest species and standard deviation of the number of individuals per species were removed from further analyses because they were highly correlated with the number of species per order and the number of individuals per species respectively. Standard deviation for mean number of species per family and for mean number of individuals per species were removed from further analyses because they were highly correlated with mean number of species per family and mean number of individuals per species respectively. Marginal tests were performed to assess the statistical significance and percentage contribution of each predictor variable alone. Models were constructed from all possible combination of variables using the BEST selection procedure. The corrected Akaike Information Criterion (AIC_c) was used to rank the models. The model with the lowest AIC_c value ($AIC_{c(min)}$) is the estimated 'best' of the candidate models. The relative importance of predictor variables were assessed and ranked by summing AIC_c model weights across all models that included that variable (Burnham and Anderson 2002). Predictor variables chosen in a model should not be interpreted as being necessarily causative as they may be acting as proxies for other important variables that either were not measured or were omitted from the model for reasons of parsimony (Anderson et al. 2008).

4.4 Results

The dataset comprised 55,395 invertebrates, 21 orders, 197 families and 751 species. The most diverse groups were Araneae (30 families, 189 species), Coleoptera (34, 140), Hemiptera (23, 126), Acarina (43, 113) and Lepidoptera (18, 56). The most abundant groups were Collembola (23,539 specimens), Acarina (8,916), Hemiptera (5,576), Araneae (5,327), Orthoptera (3,433) and Thysanoptera (3,303). More species, families and orders were collected in the ground-active invertebrate assemblages than in the foliage-active invertebrate assemblages (Table 4.2). The mean number of species per family was low (range: 2.28–2.94) with 1.29–1.25 times more species per family collected in foliage-active assemblages than in ground-active assemblages. The mean number of species per order was on average 7.2 times greater than the number of species per family. More species per order (1.39–1.68 times) were collected in foliage-active assemblages than in ground-active assemblages. More individuals per species were collected in lowland moorlands than in montane moorlands for ground-active (1.15 times) and foliage-active assemblages (1.53 times) (Table 4.2). The invertebrate assemblages surveyed had species distributions in which a few common species were most abundant (Table 4.2).

MDS ordinations of ground- and foliage-active invertebrate abundance data from lowland (Figs 4.1–4.2) and montane moorlands (Figs 4.3–4.4) indicate that as abundances are aggregated to higher taxonomic levels the overall patterns of assemblage structure are retained. Results of tests for differences between similarity matrices underlying these ordinations (Table 4.3) show that these matrices are highly correlated at all taxonomic levels. Invertebrate data aggregated at the family level were largely indistinguishable for species-level data ($\rho \geq 0.94$). Aggregation to order-level resulted in some loss of information of among site relationships ($\rho = 0.76–0.83$).

MDS ordinations (Figs 4.1–4.4) indicate that invertebrate assemblage structure in the youngest regrowth moorlands differ from older regrowth moorlands regardless of whether the assemblages were aggregated to species, family or order. ANOSIM confirmed significant differences among age of regrowth categories for ground- and foliage-active invertebrate assemblages identified to

species level and aggregated to family-level (Tables 4.4–4.5). ANOSIM also confirmed significant differences among age of regrowth categories for assemblages aggregated to order except for montane foliage-active assemblages (Tables 4.4–4.5). Global *R* decreased in size as the level of identification decreased. The pattern of pairwise *R* values among age groups was largely similar between taxonomic levels and generally decreased as the level of identification decreased. *R* values indicate that assemblage structures in the youngest regrowth differ from those in the older regrowth to a greater degree in lowland moorland than in montane moorland.

Predictors of the effectiveness of higher taxonomic surrogates

A large proportion (74%) of the variation in the effectiveness of order-level surrogates could be explained by all three predictor variables trialled (Table 4.6). There were three models within seven AIC_c units of $AIC_{c(min)}$ and thus considered to be supported by the evidence. Commonest species was the best model in terms of the least number of variables and the most variance explained (73%) and this was supported by the summed AIC_c weights. Commonest species was also highly negatively correlated with the number of species per order. There was also moderate support for the number of individuals per species and it variance.

For family-level surrogates, all five variables trialled explained a moderate proportion (42%) of the variation in higher taxonomic surrogate effectiveness (Table 4.7). The 'best' model ($AIC_{C(min)}$) comprised the number of species per family, the coefficient of variation of the number of individuals per species and commonest species and these variables received the strongest support from the summed AIC_c weights. The number of species per family is also highly correlated with its variance.

The number of species per higher taxa is negatively correlated with effectiveness and number of individuals per species and commonest species is positively correlated with effectiveness (Fig 4.5). Coefficient of variation of the number of individuals per species is negatively correlated with effectiveness at order-level and very weakly positively correlated with effectiveness at family-level (Fig. 4.5).

Table 4.2 Summary statistics for ground- and foliage-active invertebrate assemblages in lowland and montane moorlands. Values in parentheses are standard deviations. Commonest species is the number of individuals for the most common 1, 5 and 10 species divided by the total number of individuals multiplied by 100.

Statistic	Lowland		Montane	
	Ground-active	Foliage-active	Ground-active	Foliage-active
Number of sp.[1]	299	287	359	294
Number of families	131	108	145	105
Number of orders	21	12	22	13
Number of individuals	13,636	16,472	14,292	10,995
Sp./family	2.28 (2.31)	2.84 (3.67)	2.48 (2.62)	2.94 (4.43)
Sp./order	14.24 (21.22)	23.92 (26.54)	16.32 (25.12)	22.61 (28.97)
Family/sp. (ø)	0.44	0.38	0.40	0.36
Order/sp. (ø)	0.07	0.04	0.06	0.04
Individuals/sp.	45.61 (387.80)	57.39 (227.46)	39.81 (244.58)	37.40 (158.95)
Commonest sp. (%)				
1	48	13	26	17
5	70	46	56	48
10	79	64	66	68

[1]species comprises taxa identified to species or morphospecies

Table 4.3 Correlations (Spearman rank) between similarity matrices for species abundances and matrices derived from family and order abundances for ground and foliage-active assemblages in lowland and montane moorlands.*$P < 0.001$.

Level of identification	Species			
	Lowland		Montane	
	Ground-active	Foliage-active	Ground-active	Foliage-active
Family	0.94*	0.97*	0.95*	0.94*
Order	0.83*	0.82*	0.80*	0.76*

Table 4.4 ANOSIM tests for differences in lowland ground- and foliage-active invertebrate assemblages identified to species-, family and order-level among regrowth age categories (Global *R*) and pairwise tests between regrowth age categories. Regrowth age categories are identified by their mean age (years) since last burn. [a]insufficient permutations to construct a 5% significant level test. **P* < 0.05, ***P* < 0.01, ****P* < 0.001.

Test	Ground-active			Foliage-active		
	Species	Family	Order	Species	Family	Order
Global	0.38**	0.32**	0.30*	0.31*	0.33*	0.31*
3 v 13[a]	0.25	0.42	0.50	0.67	0.67	0.75
3 v 25[a]	1.00	0.82	0.85	0.89	0.89	0.96
3 v 33	0.47*	0.33	0.36	0.97**	0.96**	0.91**
3 v 57	0.90*	0.93*	0.98*	0.94*	0.89*	0.76*
13 v 25[a]	1.00	1.00	0.75	1.00	0.83	0.17
13 v 33	0.23	0.27	0.10	0.00	0.00	0.00
13 v 57[a]	1.00	0.89	0.89	0.17	–0.14	0.04
25 v 33	–0.30	–0.31	-0.30	–0.31	–0.15	0.03
25 v 57	0.28	0.20	0.35	–0.07	–0.06	–0.07
33 v 57	0.37*	0.30*	0.21	0.04	0.24	0.10

Table 4.5 ANOSIM tests for differences in montane ground- and foliage-active invertebrate assemblages identified to species-, family and order-level among regrowth age categories (Global *R*) and pairwise tests between regrowth age categories. Regrowth age categories are identified by their mean age (years) since last burn. **P* < 0.05, ***P* < 0.01, ****P* < 0.001.

Test	Ground-active			Foliage-active		
	Species	Family	Order	Species	Family	Order
Global	0.22**	0.18**	0.13*	0.28**	0.29**	0.06
2 v 6	0.07	0.04	0.05	0.16	0.21	0.10
2 v 16	0.43**	0.36**	0.30**	0.43**	0.43**	0.18*
2 v 31	0.40**	0.44**	0.20*	0.32**	0.32**	0.16*
6 v 16	0.11	0.06	0.19	0.27*	0.26*	–0.10
6 v 31	0.11	0.15	0.09	0.14	0.12	–0.06
16 v 31	0.13	0.03	-0.04	0.42**	0.32**	0.00

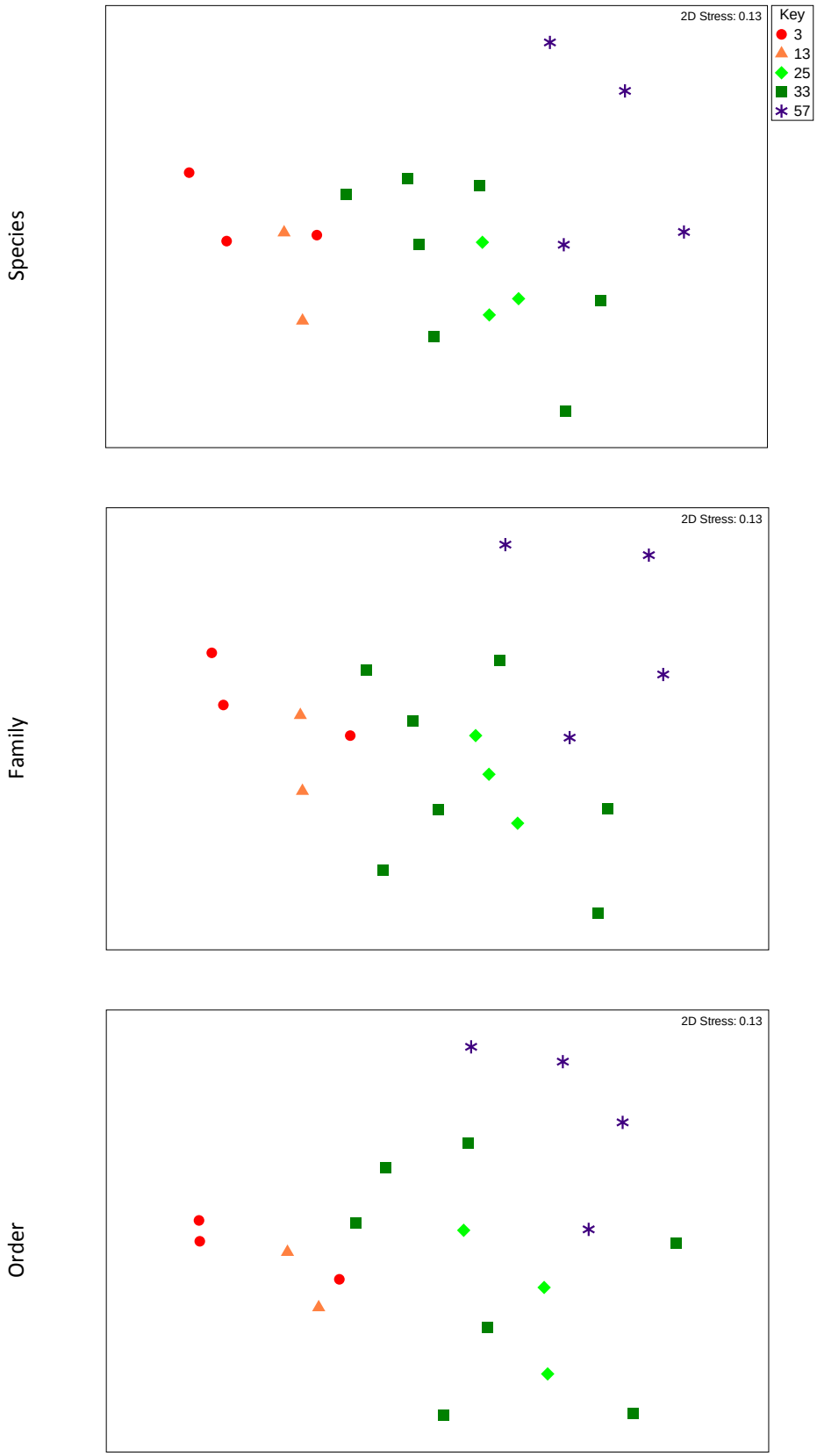

Fig. 4.1 MDS ordinations of sites in lowland moorland based on log transformed ground-active invertebrate abundances identified to species, family and order. Numbers given in the key are the mean ages (years) of the regrowth categories for sites.

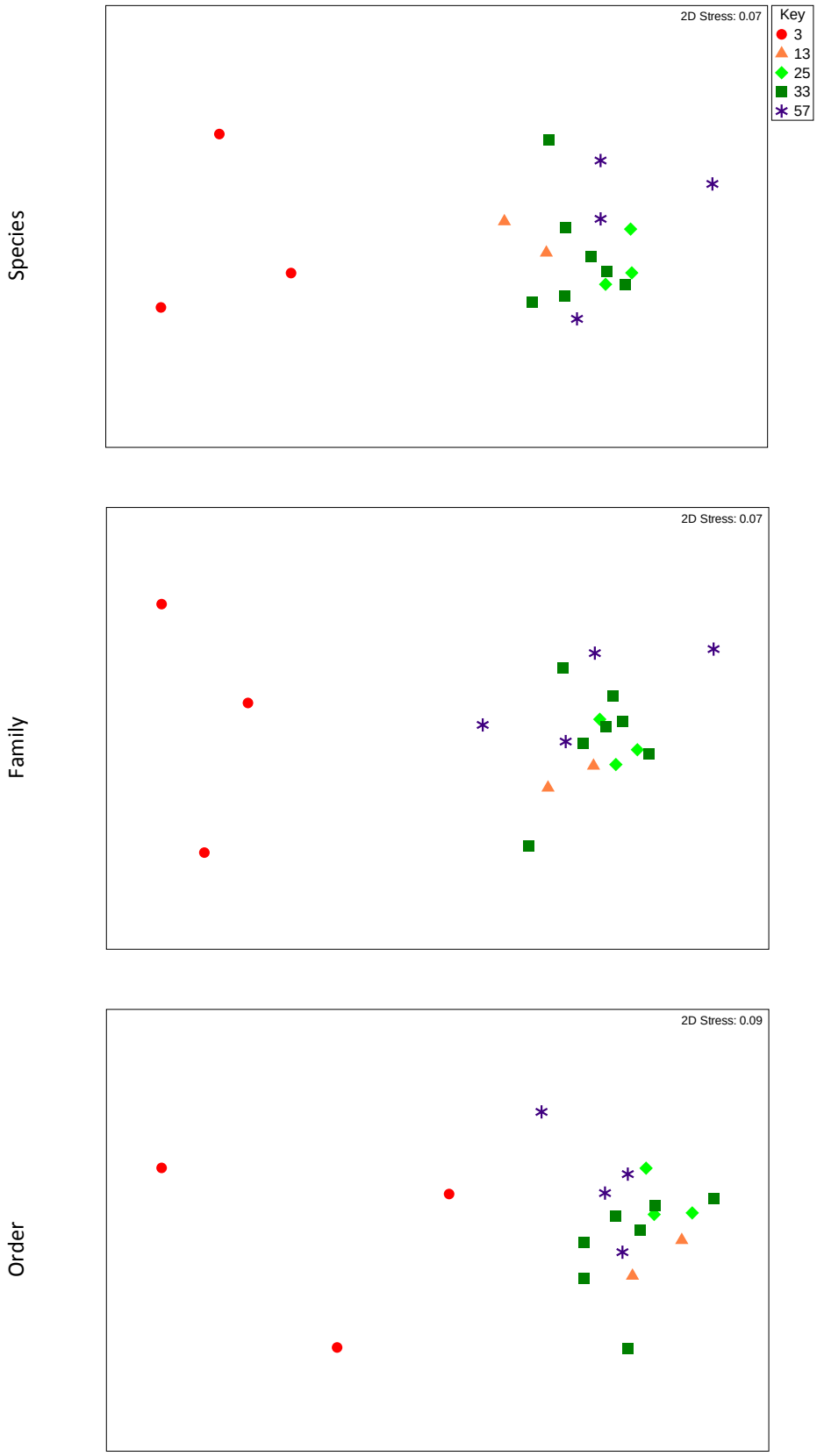

Fig. 4.2 MDS ordinations of sites in lowland moorland based on log transformed foliage-active invertebrate abundances identified to species, family and order. Numbers given in the key are the mean ages (years) of the regrowth categories for sites.

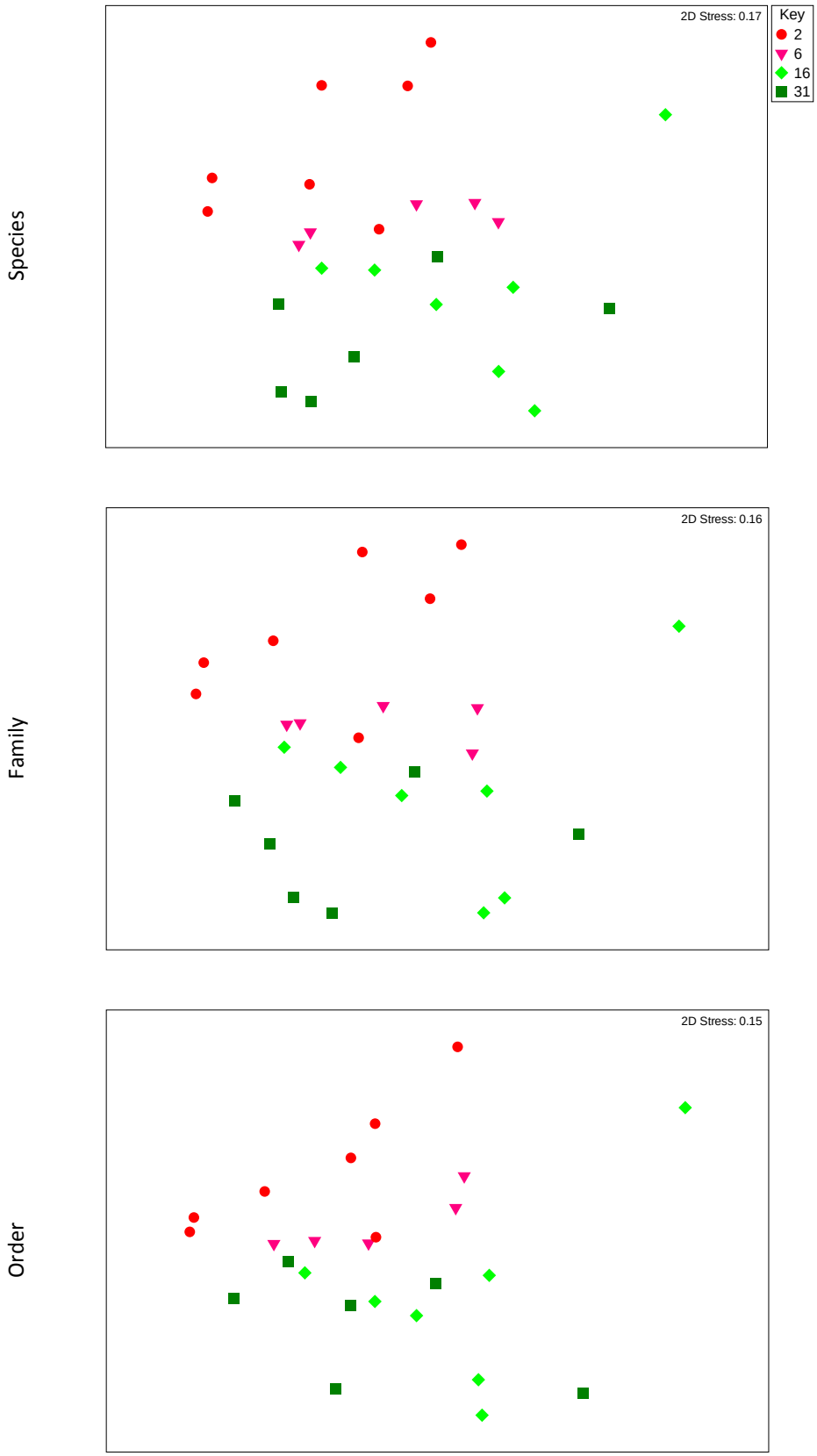

Fig. 4.3 MDS ordinations of sites in montane moorland based on log transformed ground-active invertebrate abundances identified to species, family and order. Numbers given in the key are the mean ages (years) of the regrowth categories for sites.

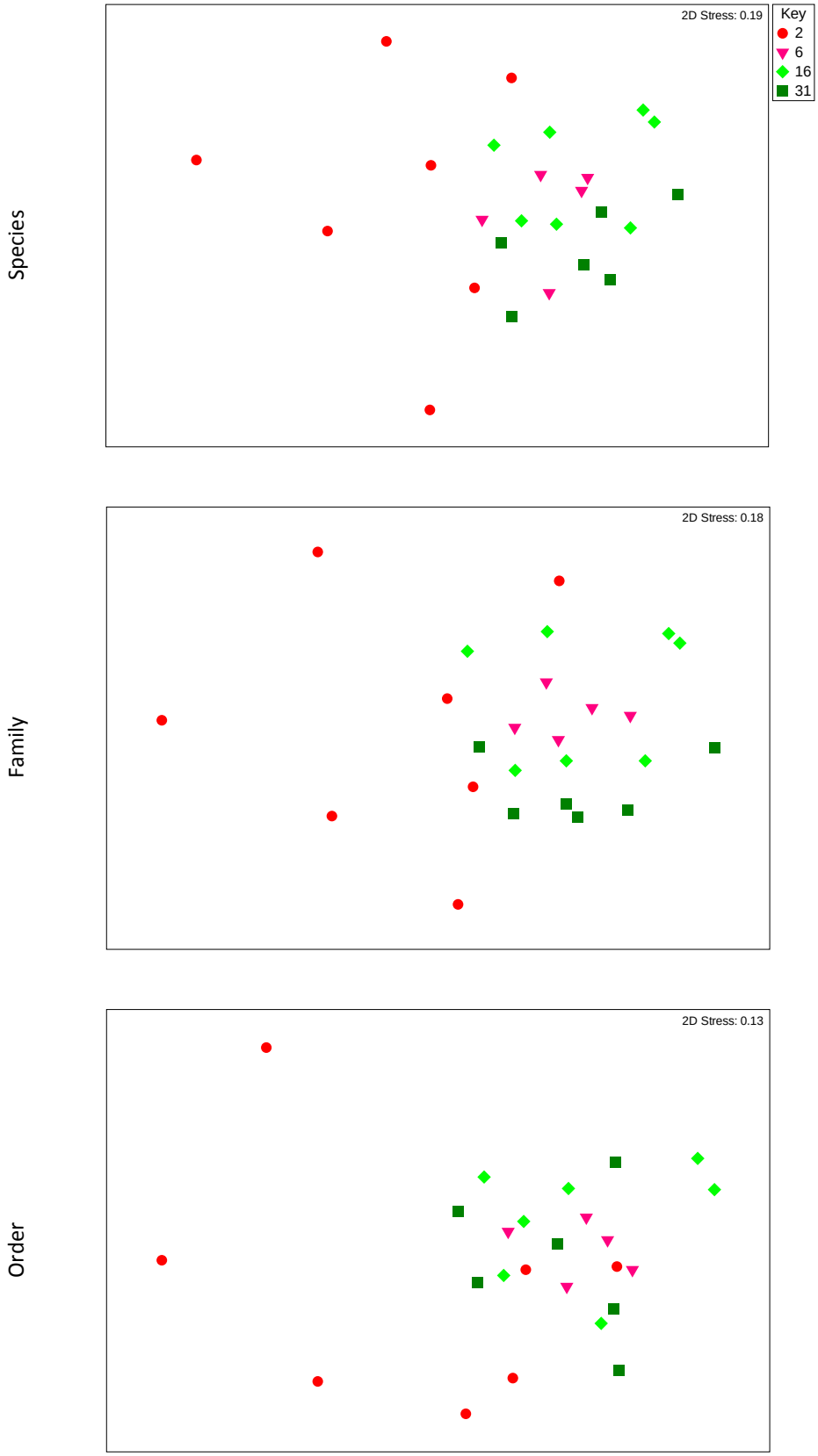

Fig. 4.4 MDS ordinations of sites in montane moorland based on log transformed foliage-active invertebrate abundances identified to species, family and order. Numbers given in the key are the mean ages (years) of the regrowth categories for sites.

Table 4.6 The contribution (R^2) of each predictor variable to explaining the variation in effectiveness of order-level surrogates and their relative importance when their AIC_c weights ($\sum\omega_i$) are summed across all DISTLM models in the multi-model approach.

Variable	Variable No.	R^2	*P*	$\sum \omega_i$
Commonest species[a]	1	0.73	0001	1.00
No. of individuals per species[b]	2	0.27	0.001	0.53
CV no. of individuals per species	3	0.05	0.123	0.27
Best models[c]				
1, 2		0.74		
1		0.73		
1, 3		0.73		
1, 2, 3		0.74		
Global model		0.74		

[a]Highly correlated with number of species per order

[b]Highly correlated with standard deviation of number of individuals per species

[c]<7 units of $AIC_{c(min)}$

Table 4.7 The contribution (R^2) of each predictor variable to explaining the variation in effectiveness of family-level surrogates and their relative importance when their AIC_c weights ($\sum\omega_i$) are summed across all DISTLM models in the multi-model approach.

Variable	Variable No.	R^2	*P*	$\sum \omega_i$
No. of species per family[a]	1	0.20	0.004	0.93
CV no. of species per family	2	0.04	0.188	0.30
No. of individuals per species[b]	3	0.22	0.001	0.38
CV no. of individuals per species	4	0.02	0.143	0.89
Commonest species (% total)	5	0.18	0.005	0.72
All models[c]				
1, 4, 5		0.41		
1, 3, 4, 5		0.41		
1, 2, 4, 5		0.41		
1, 2, 4		0.37		
1, 3, 4		0.36		
1, 4		0.31		
1, 2, 3, 4, 5		0.42		
1, 2, 3, 4		0.38		
1, 3		0.30		
1, 3, 5		0.32		
3, 4, 5		0.31		
3, 5		0.27		
4, 5		0.27		
Global model		0.42		

[1]Highly correlated with the standard deviation of the number of species per family

[2]Highly correlated with the standard deviation of the number of individuals per species

[c]< 7 units of $AIC_{c(min)}$

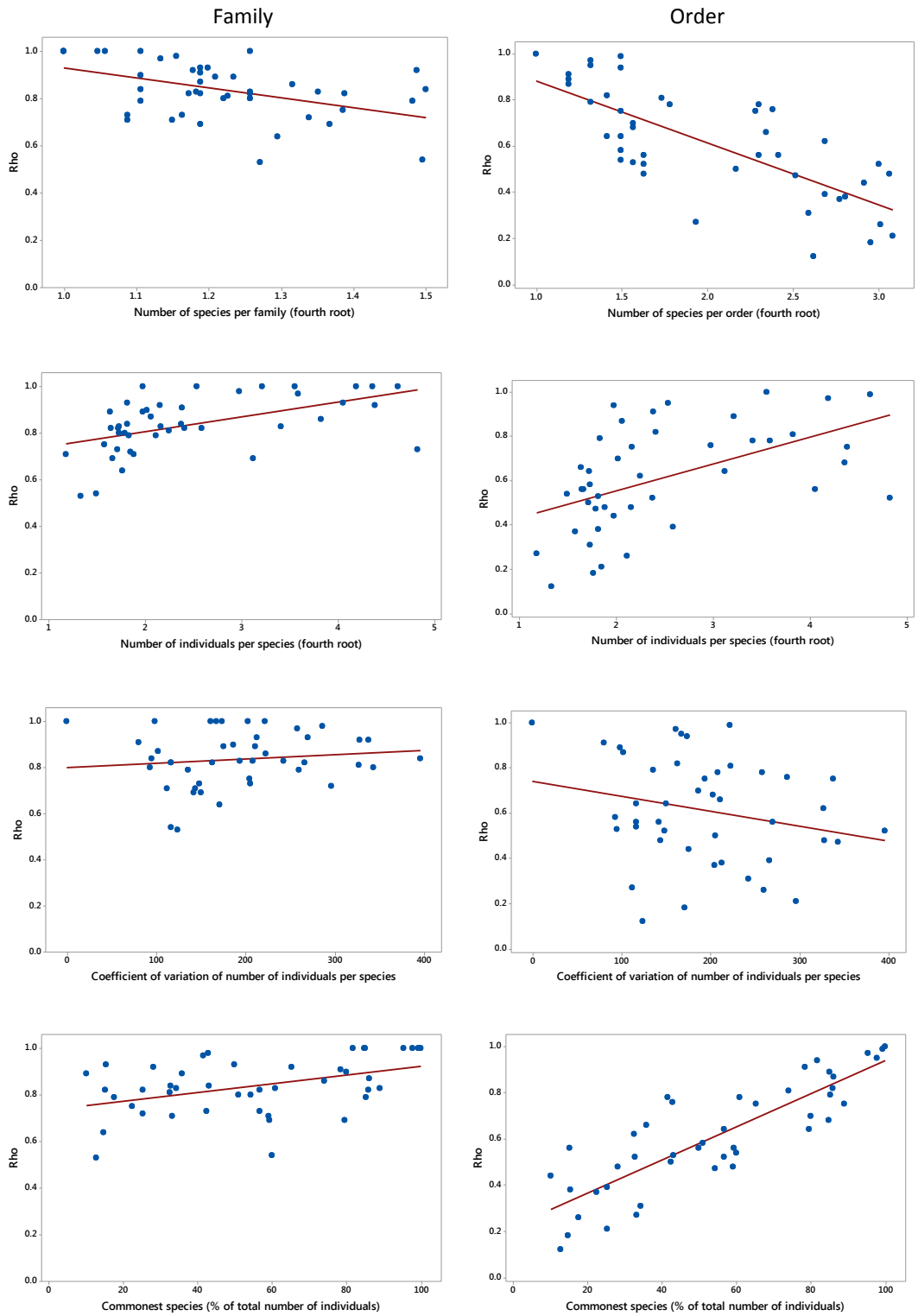

Fig. 4.5 Univariate relationships between four predictor variables and the effectiveness (Rho) of higher taxonomic surrogates (family and order).

4.5 Discussion

Family-level identification of a broad range of invertebrate groups in moorland sites with a wide range of fire history (1-60 years since last fire) was an effective surrogate for species-level identification in representing among-site assemblage patterns and detecting significant effects of regrowth age on variations in assemblage structure. This is consistent with a review and analysis of published studies by Bevilacqua et al. (2012) who found that higher taxonomic surrogates up to at least family level can be effective in depicting community responses to either natural or anthropogenic environmental variations. As expected, because of the greater ratio of number of higher taxa to number of species (Bevilacqua et al. 2012; van Rijn et al. 2015), family-level identification was a more effective surrogate than order-level identification. However, despite some loss of information, order-level resolution still provided an effective representation of among-site assemblage patterns. Only in relation to detecting effects of regrowth age on montane foliage-active invertebrate assemblage structure did order level data fail to find a similar result to species level data. This is probably due to a combination of low ratio of number of specimens to number of species, low ratio of number of orders to number of species (Table 4.1) and the limited impact of fire on montane moorland assemblages (based on Global *R* values for species and family data, Table 4.4). Several terrestrial invertebrates studies have shown that order-level identification is capable of discriminating among sites subject to differing levels of disturbance or across environmental gradients (Biaggini et al. 2007; Santos et al. 2008; Schipper et al. 2010); although, Basset et al. (2004) found poor discrimination at order level among sites along a disturbance gradient. In the marine environment where higher taxa surrogates have been more widely investigated, order-level identifications and sometimes higher taxonomic levels have been shown to be useful surrogates for species (e.g. Somerfield and Clarke 1995; Olsgard et al. 1997). The strength of perturbations is an important factor in determining whether higher taxonomic levels (i.e. above family level) are effective in detecting changes to community structure (Olsgard et al. 1998; Bevilacqua et al. 2012). This is supported in the present study because order-level identification performed better in lowland moorlands where fire had a greater impact on invertebrate assemblage structure.

However, although the effect of fire on assemblage structure was marginally stronger for foliage-active invertebrates than ground-active invertebrates (based on *R* values for species and family level data, Table 4.5), order-level analysis using ANOSIM failed to detect this effect for foliage-active invertebrates but did so for ground-active invertebrates. This suggests that other properties of the species distribution within higher taxonomic levels may be influencing the performance of higher taxa surrogates; a conclusion also reached by van Rijn et al. (2015).

Consistent with previous studies (Bevilacqua et al. 2012; Neeson et al. 2013; van Rijn et al. 2015), the number of species per higher taxa and its variance strongly influenced the performance of higher taxonomic surrogates. The smaller the ratio the better the surrogates reflected species-level assemblage patterns. I also found that surrogates performed well in communities dominated by a few common species. This has not previously been demonstrated in an empirical study; however, Neeson et al. (2013) used a theoretical model to show that higher taxonomic surrogates for species richness performed best in communities with a shallow species abundance distribution.

A criticism of higher taxonomic surrogates is that it may aggregate species within the same genus or family that have different ecological or functional traits and show different responses to disturbance (Lenat and Resh 2001; Heino and Soininen 2007; Schipper et al. 2010), and consequently limit inferences about the mechanisms by which assemblages respond to change (Somerfield and Clarke 1995; Schipper et al. 2010; Bevilacqua et al. 2013). Bevilacqua et al. (2012), working on molluscs, showed that higher taxa may be considered as arbitrary categories of species and argued that they are unlikely to convey consistent responses to disturbance. Bevilacqua et al. (2013) proposed the “Best Practicable Aggregation of Species” as an alternative approach to higher taxonomic surrogates because it does not rely on taxonomic relatedness to aggregate taxa, instead relying on ecological information from the species level matrix. Notwithstanding this criticism, higher taxonomic surrogates remain a useful, cost-effective approach for monitoring the impacts of disturbance on invertebrate biodiversity requiring limited training for non-specialists. Ultimately, their successful application will depend on the study

objectives. The level of higher taxon used will be a trade-off between time saved and the value of the results of the survey.

4.6 Conclusions

Using a broad range of moorland invertebrates (>20 orders), this study has shown that family-level identification is an effective surrogate for species-level identification in representing patterns in assemblage structure and detecting the effects of fire. Order-level identification can also be an effective surrogate but the level of discrimination among sites will typically be lower than species- or family-level identification, and it may not detect more subtle changes as a result of disturbance. Higher taxonomic surrogates performed well for invertebrate orders comprising a few species that were abundant and with a small mean and variance in the number of species per higher taxa.

Plate 19 Silver xenica *Oreixenica lathoniella* (Westwood) (Lepidoptera: Nymphalidae)

Plate 20 Tasmanian black cicada *Diemeniana tillyardi* Hardy (Hemiptera: Auchenorrhyncha: Cicadidae)

Plate 21 Collembola *Corynephoria* sp. (Collembola: Bourletiellidae)

Chapter 5

Invertebrate compositional changes caused by low-intensity fire in moorland can be predicted by changes in environmental variables: evidence from a before-after-control-impact approach

5.1 Abstract

Understanding the effects of fire on invertebrates and the factors that predict changes is important for effective ecosystem management. The responses of ground- and foliage-active invertebrates to low-intensity fire at two moorland sites (low and moderate productivity) were investigated using a before-after-control-impact design. Higher taxa were used as surrogates for species-level identification. The composition of invertebrate assemblages changed in response to fire, with most taxa less abundant 1–2 years after fire than before fire. Although the assemblages had largely returned to the pre-fire state within six years after fire, the composition of foliage-active invertebrate assemblages at the low productivity site differed between burned and unburned plots 14 years after fire. Contrary to expectations, there was only a weak effect of fire on ground-active invertebrates at the low productivity site. Vegetation density and cover of thatch were found to be strong predictors of variation in invertebrate assemblage composition. The short–medium-term changes in composition of ground- and foliage-active invertebrates, with no loss of invertebrate groups, indicate that, at higher taxonomic levels, these components of the invertebrate fauna have considerable resilience to single fires.

Key words: arthropods, higher taxonomic surrogates, soil productivity, Tasmania

5.2 Introduction

Fire is an integral part of ecosystem management in many biomes around the world. This is particularly the case for the fire-prone and highly flammable moorlands of western Tasmania. These moorlands were burnt by hunter-gathers over thousands of years to facilitate easy passage and to encourage game (Thomas

1993), and it has been inferred that they conducted frequent, low-intensity fires (Marsden-Smedley and Kirkpatrick 2000). Following the displacement of Indigenous people from Tasmania by Europeans in the nineteenth century, the fire incidence immediately reduced (von Platen et al. 2011). However, in inland areas this was followed by an increased incidence of fire, including large-scale fires, caused by Europeans expediting passage in their quest for resources, which also burnt extensive areas of fire-sensitive rainforest (Marsden-Smedley 1998; Johnson and Marsden-Smedley 2001). In coastal areas that were less used by people after displacement of the Indigenes, the fire incidence decreased, resulting in changes in the distribution of vegetation types (di Folco and Kirkpatrick 2013). Increased burning in moorlands of western Tasmania has been advocated for the benefit of the species that depend on this habitat, and to reduce the risk of catastrophic fires in adjacent fire-sensitive vegetation (Marsden-Smedley 1998; Marsden-Smedley and Kirkpatrick 2000). Although moorland vegetation has a well-developed capacity to return to the pre-fire state following infrequent fire, either vegetatively or from seed (Balmer and Storey 2010), considerably less is known about the capacity of fauna to do the same; especially invertebrates which comprise a significant proportion of the biodiversity in this community.

Few studies have investigated the response of invertebrates to fire in Tasmanian moorlands (Greenslade and Driessen 1999; Driessen and Greenslade 2004; Green 2009; Chapter 2; Chapter 3) and most had scope and design limitations. These studies found that fire typically has a short-term effect on invertebrate assemblage composition and that return to the pre-fire state correlates with the return of vegetation and other habitat components to the pre-fire state. This pattern of response to fire is similar to many other studies in fire-prone ecosystems (Friend and Williams 1996; Andersen and Müller 2000; Bess et al. 2002; Parr et al. 2004; Brennan et al. 2006; Pyrke and Samways 2012a). However, in contrast to the general pattern of response following fire in fire-prone communities, ground-active invertebrate abundance was greater in young regrowth moorland than in older regrowth owing to large numbers of Formicidae, Orthoptera, Collembola and Diptera (Chapter 3). Furthermore, Green (2009) found that both the density and diversity of soil-active Acarina were significantly higher in 30–40 year

old regrowth moorland compared with younger regrowth, suggesting that these Acarina populations take a long time to return to high levels of density and diversity following fire.

Previous studies of invertebrate responses to fire in Tasmanian moorlands have used space as surrogates for time, as have many other fire studies around the world. This study design can provide useful insights into long-term effects of fire on biodiversity (Clarke 2008; Driscoll et al. 2010) but has also been criticised for its potential to lead to misinterpretations (Johnson and Miyanishi 2008). Combining long-term chronosequences with experimental designs has been suggested as a way to improve insights about community and ecosystem change (Walker et al. 2010). Here, a more robust before-after-control-impact design is used to assess the short–medium-term (1–14 years) effect of low-intensity fire on ground- and foliage-active invertebrates in low and moderate productivity moorland sites. Based on previous studies on a broad range of invertebrate groups in fire-prone ecosystems (e.g. Moretti et al. 2004; Pyrke and Samways 2012a), it is expected that fire will result in change in the local environment (e.g. vegetation biomass, structural complexity, litter and decaying vegetation) and this will result in short-term changes (at least one year after fire) in assemblage composition and a decrease in invertebrate abundance and diversity. It is expected that the invertebrate assemblages will increasingly resemble the corresponding assemblages in unburnt controls as environments return to the pre-fire state. It is also expected that both environment and invertebrate assemblages will return to the pre-fire state more quickly after fire on higher productivity sites.

In the present study, invertebrates were identified predominantly to the level of order, because the cost of identifying them to the level of family or species was prohibitive. Previous research suggests that higher taxa (order level) are effective surrogates for species-level data in representing patterns of assemblage composition due to fire disturbance in buttongrass moorlands; however, subtle changes may not be detected (Chapter 4).

This chapter addresses the hypotheses that (i) the environmental changes caused by fire in moorland are reflected in the invertebrate fauna and (ii) the process of return to pre-fire conditions will be more rapid on more productive sites.

5.3 Methods

Study sites

The study was conducted at two sites (80 km apart) in western Tasmania: a lowland moorland site (320 m above sea level) at Airstrip Road (42° 51' S, 146° 14' E) and a montane moorland site (780 m above sea level) at King William Creek (42° 12' S, 146° 08' E). Based on nearby weather stations (at Lake St Clair and Strathgordon), both sites have similar mean annual rainfall (lowland, 1,951 mm; montane, 1,868 mm,) and monthly rainfall patterns with rainfall highest in winter and lowest in January–March. Minimum temperatures are 2–3°C lower throughout the year at the montane site (February, 7.3°C; July, –0.1°C) than at the lowland site (February, 9.2°C; July, 3.2°C). During summer, maximum temperatures are similar at the two sites (19–20°C) but cooler during spring and autumn and up to 4°C cooler during winter at the montane site (6.2°C) than at the lowland site (10.0°C).

Moorland vegetation differed between the two sites with each comprising one of the two major groups of buttongrass recognised by Jarman et al. (1988). Moorland at the lowland site is classified as "standard blanket moor", which is widespread in lowland areas of western Tasmania, occurring within a wide range of environmental situations on ancient sediments that carry shallow infertile soils (Jackson 1968). The lowland site was last burnt in a wildfire in 1972; 27 years before sampling in the present study commenced. Moorland at the montane site is classified as "common highland sedgey", which is widespread in highland regions of western and central Tasmania where it is associated with dolerite geology with relatively fertile soils (Jackson 1968). The montane site was last burnt in 1987 by a low-intensity fuel reduction burn; 12 years prior to the commencement of sampling in the present study. Moorlands at both sites are dominated by sedges; however, the lowland moorland has a conspicuous shrub component whereas the montane moorland has a very sparse shrub component. Grasses and herbs are more common in the montane moorland than in the lowland moorland.

Experimental design and data collection

A before-after-control-impact design was used to investigate the responses of invertebrate assemblages to low intensity fire. At each site six blocks were

established with three blocks randomly allocated as treatment blocks (burn) and three as control blocks (no burn). These were surveyed once a year in late summer (February–March) to maximise species richness and abundance and to enable comparison with the results from the space-for-time study (Chapter 3). The lowland blocks (30 by 30 m) were arranged in a line and were surveyed on three occasions, spanning three years, prior to the treatment burns and then on six occasions, spanning 1–14 years, after the burns. The montane blocks (50 by 50 m) were arranged in a two by three grid and were sampled on one occasion prior to the treatment burns and then on six occasions, spanning 1–7 years, after the burns. The longer period of post-burn monitoring at the lowland site was because other studies have shown that lowland moorlands take longer to re-establish than montane moorlands (Chapters 2–3). The low-intensity (<500 kW m^{-1}) treatment burns were conducted in autumn and the plains surrounding the control blocks were also burnt.

Pitfall traps and sweep nets were used to sample ground- and foliage-active invertebrates respectively. A pitfall trap comprised a plastic 225-ml drinking cup with a 7-cm opening inserted into a PVC tube, with a roof to prevent rain from entering. It contained 100 ml of 70% ethanol and a dash of glycerol. Sixteen pitfall traps were set in a four-by-four arrangement (4–5 m between traps) centered in the middle of each block. Traps were not placed in hollows where they were likely to be flooded by surface run-off. Pitfall traps operated for two weeks but were cleared and reset after one week. The sweep net comprised a 45.6-cm diameter hoop, 71-cm handle and a white funnel tapered net that was 81 cm long with a 0.9- by 0.3-mm mesh size. A sweep sample comprised 100 passes of the net across the top of vegetation. One sweep of the net would occur at every pace of the collector. Four sweep samples were taken each survey, two each on separate days. A sweep sample would involve three traverses of the block at the lowland site and two traverses at the montane site. To ensure the same part of the block was not swept more than once within a survey, imaginary lines across the block were identified and randomly selected for each sweep sample. At the completion of each sample, all large twigs that were collected in the net were beaten against the inside of the net and removed. Invertebrates in the net were transferred into a sample jar (50 by 50 by 80 cm) containing 70% ethanol. Sweep samples were performed between

10:00 a.m. and 6:00 p.m. on fine days, with the temperature ranging between 18 and 25°C.

Invertebrates were sorted to major taxonomic groups (predominantly order) by using a dissecting microscope at 10X magnification, and the number of specimens were counted and stored in separate vials for each pitfall and sweep sample. The predominant collections of holometabolous and hemimetabolous insects were adults. Larvae of Lepidoptera, Diptera, Hymenoptera and Coleoptera were identified and counted separately from adults because they have markedly different ecological roles. For taxa with similar-looking immatures and adults (e.g. arachnids, paurometabolous insects) these life stages were counted together. Formicidae were counted separately from other Hymenoptera because ants are widely studied as indicator insects responsive to environmental change.

Pitfall trapping and sweep sampling contribute only a partial assessment of the entire insect population in buttongrass moorlands and the number of individuals and life stages caught by these methods reflects both activity and abundance of the sampled invertebrates. For convenience, the term abundance is used here when referring to the number of invertebrates caught by these sampling methods.

Vegetation height and vegetation density was measured at eight pitfall trap positions per block (two opposite outer rows) using a 100-cm-tall by 50-cm-wide board marked with 200 5- by 5-cm squares. The board was placed on the ground behind each pitfall trap on the two outer rows and viewed by a crouched observer looking across the block from corresponding pitfall traps 10 m away—this ensured consistent observations between years. The index of vegetation density was the number of squares obscured by vegetation. Vegetation height (cm) was the height that had at least five squares in a row obscured by vegetation.

For every second pitfall trap station, the percentage cover in a 2- by 2-m quadrat was estimated for: shrubs, grasses, buttongrass, sedges (excluding buttongrass and including rushes), herbs, cryptograms, ferns, bare ground, litter (detached plant material), and dead plant material (standing/attached dead plant material). At the montane site, during the first year only, vegetation cover was recorded using the modified Braun-Blanquet cover index (0.5, < 1%; 1, 1–5%; 2, 6–

25%; 3, 26–50%; 4, 51–75%; 5, 76–100%) and this was converted to percentage cover values by taking the midpoint of the Braun-Blanquet percentage cover ranges. Percentage vegetation cover values were not recorded for all years that invertebrates were surveyed. Vegetation cover data was recorded by DPIPWE botanical staff as part of their investigation into the effects of fire on vegetation and is used here with their kind permission.

Data analysis

The design consisted of two factors: Treatment (T; fixed, 2 levels: burn and unburnt control) and Time Since Treatment Burn (TSTB; random, invertebrates: 9 levels for the lowland site, 7 levels for the montane site; vegetation: 5 levels), with n = 3 replicate observations per combination of factors obtained from the mean of 16 pitfall traps, four sweep samples and eight vegetation plots per block. The number of variables used for each multivariate analysis was 32 for lowland ground-active invertebrates, 26 for lowland-foliage-active invertebrates, 35 for montane ground-active invertebrates, 23 for montane foliage-active invertebrates, eight for lowland vegetation and nine for montane vegetation. Permutational distance-based multivariate analysis of variance (ANOVA), PERMANOVA (Anderson 2001; Anderson et al. 2008) was used to analyse the full designs, using 4,999 random permutations under a reduced model. Analyses were based on Bray–Curtis dissimilarities on fourth root transformed data for invertebrates and square root transformed data for vegetation. Where there was a significant interaction between TSTB and T, pairwise comparisons between burn and control blocks within year were undertaken. As there were too few possible permutations to obtain a reasonable test, a *P*-value was calculated using 4,999 Monte Carlo draws from the appropriated asymptotic permutation distribution (Anderson and Robinson 2003). Nonmetric multidimensional scaling was used to visualise relationships among treatments through time.

In addition to PERMANOVA, non-parametric, one-way analyses of similarity (ANOSIM) were performed on invertebrate and vegetation data to determine how similar assemblage compositions were between burn and control treatments for each TSTB. ANOSIM returns an *R*-statistic which gives a measure of how similar

categories are. *R* values commonly range from 0 to 1; the close the *R*-value is to 1 the more different the categories are, while a value close to zero indicates assemblages can barely be separated (Clarke and Warwick 2001).

The RELATE routine was used to test if among-sample relationship assemblages were similar to the among-sample relationships for cover composition.

The distance-based linear model (DISTLM) routine (Anderson et al. 2008) was used to explore and model the relationship between the invertebrate assemblage composition and environmental variables. Prior to analyses, draftsman's plots were examined to determine if the samples were roughly symmetrically distributed across the range of each variable. Cover of litter at the lowland site and cover of bare ground and herbs at the montane site were square root transformed. Where a variable was highly correlated (r >|0.8|) with another variable one was removed from the analyses. This reduced the number of variables to a reasonable number for the size of the data sets (Anderson and Burnhan 2002).

Marginal tests were performed to assess the statistical significance and percentage contribution of each environmental variable alone. Models were constructed from all possible combinations of environmental variables using the BEST selection procedure. The corrected Akaike Information Criterion (AIC_c) was used to rank the models. The model with the lowest AIC_c value ($AIC_{c(min)}$) is the estimated 'best' of the candidate models. Because models within seven AIC_c units of ($AIC_{c(min)}$) have some support and should rarely be dismissed (Burnham et al. 2011), the relative importance of predictor variables were assessed and ranked by summing AIC_c model weights across all models that included that variable (Burnham and Anderson 2002). Predictor variables chosen in a model should not be interpreted as being necessarily causative as they may be acting as proxies for other important variables that either were not measured or were omitted from the model for reasons of parsimony (Anderson et al. 2008).

The BIOENV routine was used to detect any variables that the linear DISTLM approach may have missed (Clarke and Gorley 2006; Hallet et al. 2012). BIOENV is a less constrained, fully non-parametric method which caters for non-linear functions (Clarke and Ainsworth 1993) and was used to find the best matches between the among-sample patterns for the invertebrate assemblage patterns and those from

the environmental variables associated with those samples (Clarke and Gorley 2006). The null hypothesis of no similarities in rank order pattern between the complementary matrices was rejected if the significance level (*P*) associated with the test statistic (Spearman's rank 'matrix correlation' coefficient [ρ]) was < 0.05. The extent of any significant differences was determined by the magnitude of ρ, with values close to zero indicating little correlation in rank order pattern whereas values close to 1 indicated a near perfect agreement. R^2 and ρ are broadly comparable since the latter is a matrix correlation, not a direct correlation (Hallet et al. 2012).

Analyses of variance were used to examine the effects of fire on total invertebrate abundance, number of invertebrate groups and the abundance of individual taxa. The model of analysis was the same as that described for the multivariate case. Where there was a significant interaction between TSTB and T, pairwise comparisons between burn and control blocks within year were undertaken. Analyses of variance were performed in PERMANOVA, with *P*-values obtained using 4,999 random permutations for the main test and 4,999 Monte Carlo draws from the appropriated asymptotic permutation distribution for the pairwise tests. To reduce the probability of falsely accepting null hypotheses (type II errors), alpha was set at 0.1 for the main test. However, for pairwise comparisons alpha was set at 0.05.

5.4 Results

A total of 290,077 invertebrates were collected representing 36 orders or classes. The most abundant taxa accounting for 95% of specimens collected were Collembola (110,163 individuals), Diptera (44,704), Acarina (34,579), Hemiptera (27,787), Araneae (24,194), Hymenoptera (15,064), Orthoptera (10,978) and Thysanoptera (6,711).

Changes in total invertebrate abundance and number of invertebrate groups

Both total invertebrate abundance and the number of invertebrate groups were significantly less one year after treatment burns at both sites than before the burns, except for montane ground-active invertebrates (Table 5.1, Figs 5.1–5.2).

For lowland foliage-active invertebrates, abundance was lower on burnt blocks than unburnt blocks for up to six years post-fire and the number of groups was lower on burnt blocks than unburnt blocks for up to 14 years post-fire. In contrast, montane foliage-active invertebrates generally returned to the pre-fire state two years post-fire; although significantly less invertebrate groups and numbers of invertebrates were detected on burnt blocks three and seven years post-fire respectively.

Abundance of ground-active invertebrates returned to the pre-fire state two years post-fire at both sites; however, at the lowland site there was a significant increase in invertebrate abundance on burnt blocks three years post-fire (Fig. 5.1). This post-fire increase in invertebrate abundance was a result of significant increases in numbers of Collembola and Hemiptera and non-significant increases in numbers of Diptera, Hymenoptera and Orthoptera (Fig. 5.3).

Most invertebrate groups had a significant treatment by TSTB interaction effect on abundance and the majority of these groups had reduced abundance from the pre-fire condition at least in the first year following the treatment burn (Figs 5.3–4). This effect of fire lasted 4–6 years for foliage-active Thysanoptera and Hemiptera (lowland) and ground-active Amphipoda (montane), and 14 years post-fire for lowland foliage-active Collembola. The only taxon that increased following treatment burns was the Orthoptera; 1–4 years post-fire in montane ground samples and 1–2 years post-fire in lowland foliage samples. At the montane site, foliage-active Lepidoptera larvae were not caught on burned or unburned blocks one year post-fire. In the following year significantly higher numbers of Lepidoptera larvae were caught on the burned than on the unburned blocks (Figs 5.3–4).

Compositional changes in invertebrate assemblages

There were significant interaction effects of treatment and TSTB on the composition of invertebrates at both sites (Table 5.2). Ordinations clearly show these shifts in invertebrate assemblage composition except for lowland ground-active invertebrates (Fig. 5.5). Prior to the burns, there were no significant differences in invertebrate assemblage composition between control and treatment blocks (Table 5.3, Fig. 5.5). Post treatment, there were strong differences (ANOSIM $R > 0.7$) in

assemblage composition between control and treatment blocks for foliage-active invertebrates at both sites and ground-active invertebrates at the montane site. At the montane site, these differences in composition remained detectable five years post-fire for ground-active invertebrates and at least four years for foliage-active invertebrates—ANOSIM *R* values suggest weak–moderate differences occur up to seven years post-fire. At the lowland site, strong differences in foliage-active assemblage composition between control and treatment blocks remained detectable 14 years post-fire at the conclusion of sampling (ANOSIM *R* = 0.70, Table 5.3). For lowland ground-active invertebrates there were only weak–moderate differences in assemblage composition between treatment and control blocks for up to six years following fire (ANOSIM *R* = 0.26–0.44, Table 5.3).

Compositional change in vegetation

At both sites there were significant interaction effects of treatment and TSTB on vegetation composition (Table 5.4, Fig 5.5). Vegetation composition did not differ significantly between treatment and control blocks prior to burns. Post treatment there were significant differences in vegetation composition between the burned and unburned blocks at both sites (Table 5.5, Fig. 5.5). At the lowland site, significant differences lasted for at least six years post-fire. At the montane site, strong differences in vegetation composition (ANOSIM $R > 0.8$) lasted three years post-fire.

At the montane site there was some evidence to suggest moderate differences in vegetation composition between treatment and control blocks prior to the treatment burn (Table 5.5); however, seven years post-treatment burn there was little evidence to suggest a difference between treatment and control blocks. This may suggest that the vegetation composition has changed from the pre-burn state or it may be an artefact of the change of recording from Braun-Blanquet to percentage cover.

There were significant interaction effects of treatment and TSTB for both vegetation density (lowland: Pseudo $F_{(1,4)} = 31.852$, $P_{(perm)} = 0.0002$, montane: Pseudo $F_{(1,4)} = 5.699$, $P_{(perm)} = 0.0036$) and litter cover (lowland: Pseudo $F_{(1,4)} = 3.800$, $P_{(perm)} = 0.0148$; montane: Pseudo $F_{(1,4)} = 23.816$, $P_{(perm)} = 0.0002$). At both sites

vegetation density decreased markedly following fire, with return to pre-fire levels occurring by three years at the montane site on moderate productivity soils (Fig. 5.6). At the lowland site on low productivity soils, vegetation density had not returned to pre-fire levels 14 years post-fire (Fig. 5.6). At both sites litter cover increased following fire due to the presence of thatch and this increase lasted at least two years but less than 5–6 years (Figs. 5.7).

Environmental predictors of invertebrate assemblage composition

The RELATE analysis indicated that the among-sample relationships based on invertebrate assemblage compositions were moderately related to the among-sample composition based on vegetation composition at both sites: lowland, ground-active invertebrates (ANOSIM R = 0.401, P = 0.02); lowland, foliage-active invertebrates (R = 0.560, P = 0.02); montane, ground-active invertebrates (R = 0.461, P = 0.02); montane, foliage-active invertebrates (R = 0.499, P = 0.02).

All environmental predictor variables together explained large proportions (48–68%) of the variation in the composition of invertebrate assemblages (Tables 5.6–5.7). There were a large number of models within seven AIC_c units of $AIC_{c(min)}$ (lowland, 155–227; montane, 273–680). Summed weights across all models were therefore used to indicate the relative importance of each variable. There was moderate–strong support for litter cover and vegetation density as predictors of invertebrate assemblage composition at both sites, with stronger support for these two predictor variables for foliage-active invertebrates than for ground-active invertebrates. For ground-active invertebrates at both sites there was moderate support for several predictor variables such as cover of buttongrass, herbs, shrubs and cryptograms (lowland site) and cover of bare ground, grasses and herbs and regrowth age (montane site).

The BIOENV routine also identified vegetation density as a potentially important predictor of invertebrate assemblage composition either with litter cover (lowland invertebrates, montane foliage-active invertebrates) or bare ground (montane invertebrates), or alone (Table 5.8).

Table 5.1 PERMANOVA examining effects of treatment burns on the abundance and number of groups of ground- and foliage-active invertebrates for lowland and montane moorlands. Each test was done using 4,999 permutations. Sq. root is the square toot of the component of variation attributable to that factor in the model, in units of Euclidean distance. TSTB, time since treatment burn.

	Number of invertebrates						Number of invertebrate groups					
Source	df	SS	MS	Pseud-F	$P_{(perm)}$	Sq.root	df	SS	MS	Pseud-F	$P_{(perm)}$	Sq.root
Lowland site												
Ground-active												
Burn	1	648.98	648.98	0.63	0.4524	-3.75	1	0.11	0.11	0.29	0.6160	-0.10
TSTB	8	32499.00	4062.30	23.36	0.0002	25.46	8	179.71	22.46	161.38	0.0002	1.93
Burn X TSTB	8	8227.30	1028.40	5.91	0.0002	16.88	8	2.97	0.37	2.67	0.0190	0.28
Residual	36	6260.60	173.91			13.19	36	5.01	0.14			0.37
Total	53	47635.00					53	187.80				
Foliage-active												
Burn	1	130980.00	130980.00	3.22	0.1150	57.84	1	22.36	22.36	9.09	0.0190	0.86
TSTB	8	2274400.00	284300.00	32.56	0.0002	214.31	8	61.37	7.67	12.14	0.0002	1.08
Burn X TSTB	8	325210.00	40651.00	4.66	0.0004	103.15	8	19.69	2.46	3.89	0.0010	0.78
Residual	36	314320.00	8731.10			93.44	36	22.75	0.63			0.79
Total	53	3044900.00					53	126.17				
Montane site												
Ground-active												

Burn	1	8.53	8.53	0.06	0.8304	-2.64	1	0.14	0.14	0.32	0.5978	-0.12
TSTB	6	1902.20	317.04	5.09	0.0012	6.52	6	7.66	1.28	3.47	0.0120	0.39
Burn X TSTB	6	929.94	154.99	2.49	0.0456	5.56	6	2.58	0.43	1.17	0.3478	0.14
Residual	28	1743.90	62.28			7.89	28	10.29	0.37			0.61
Total	41	4584.60					41	20.66				
Foliage-active												
Burn	1	48904.00	48904.00	4.68	0.0782	42.79	1	1.89	1.89	1.77	0.2246	0.20
TSTB	6	428470.00	71411.00	35.38	0.0002	107.54	6	77.27	12.88	34.73	0.0002	1.44
Burn X TSTB	6	62736.00	10456.00	5.18	0.0014	53.03	6	6.41	1.07	2.88	0.0240	0.48
Residual	28	56518.00	2018.50			44.93	28	10.38	0.37			0.61
Total	41	596630.00					41	95.96				

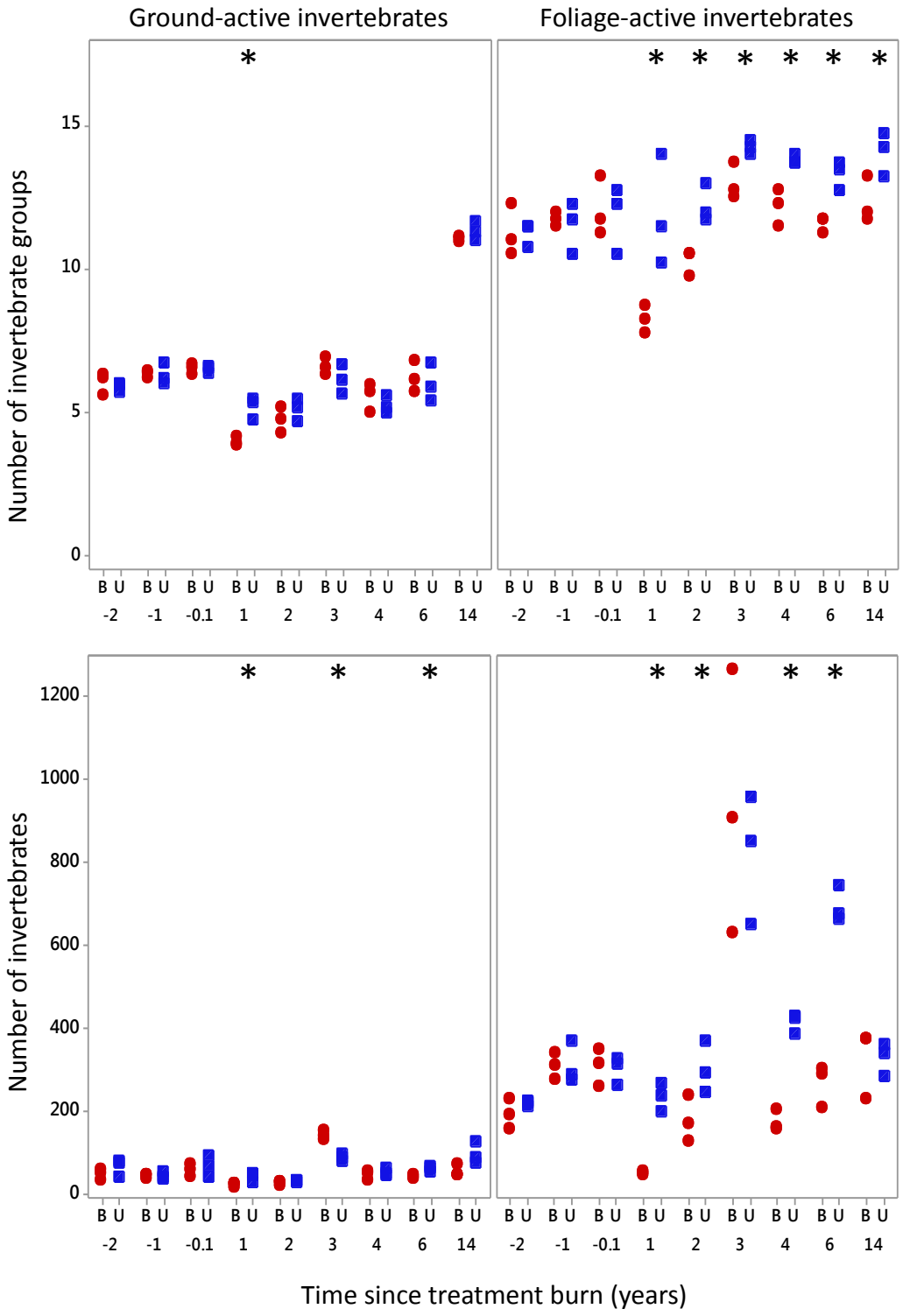

Fig. 5.1 Individual value plots of the number of invertebrate groups and the number of invertebrates for three burned blocks (red circles) and three unburned blocks (blue squares) for each time-since-treatment-burn at the lowland site. Each block value is based on the average of 16 pitfall traps for ground-active invertebrates and four sweep samples for foliage-active invertebrate. B, burned; U, unburned; *significant pairwise differences ($P < 0.05$) between treatments for global tests with significant time since treatment burn and treatment interaction.

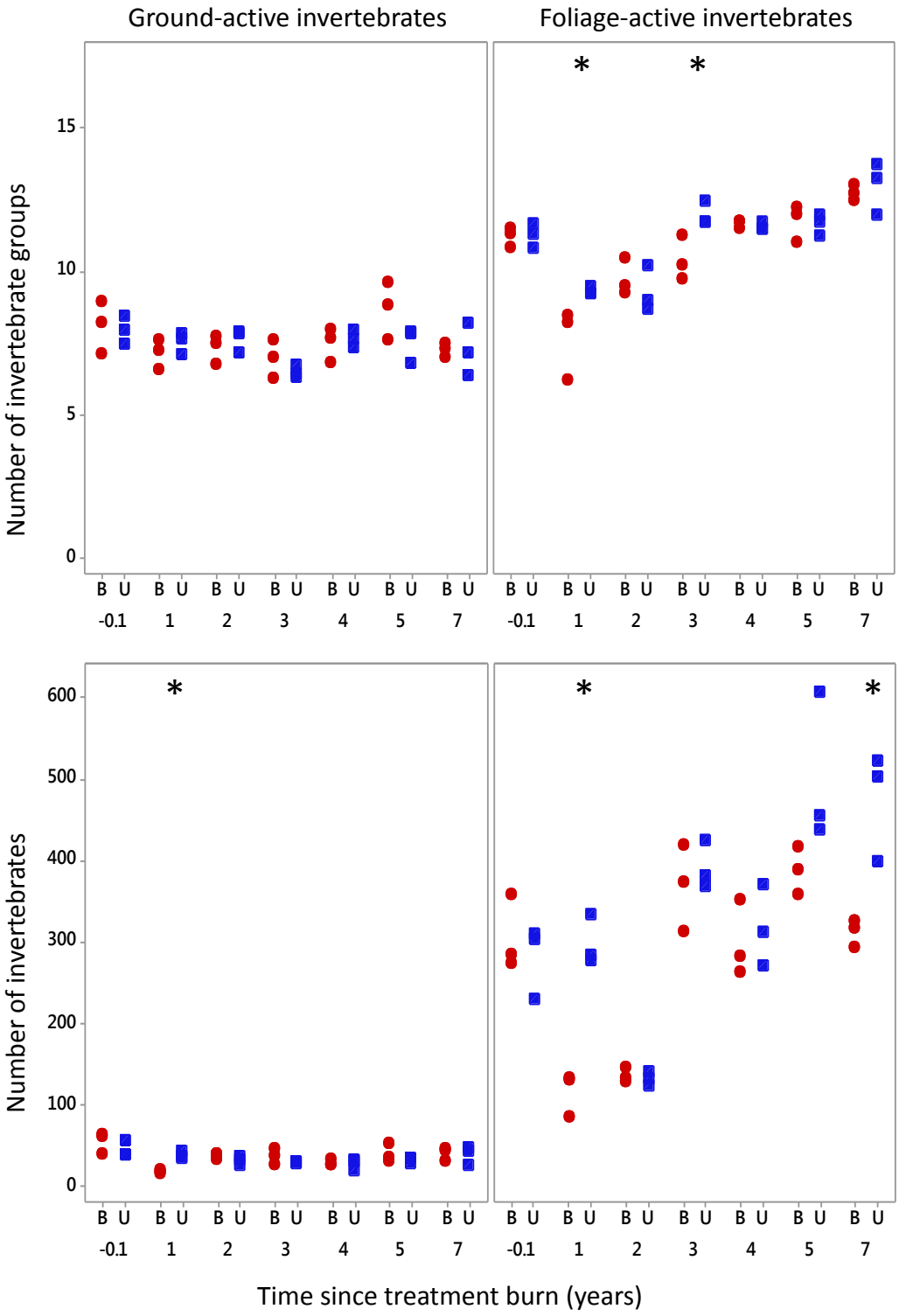

Fig. 5.2 Individual value plots of the number of invertebrate groups and number of invertebrates for three burned blocks (red circles) and three unburned blocks (blue squares) for each time-since-treatment-burn at the montane site. Each block value is based on the average of 16 pitfall traps for ground-active invertebrates and four sweep samples for foliage-active invertebrate. B, burned; U, unburned; *significant pairwise differences (P < 0.05) between treatments for global tests with significant time since treatment burn and treatment interaction.

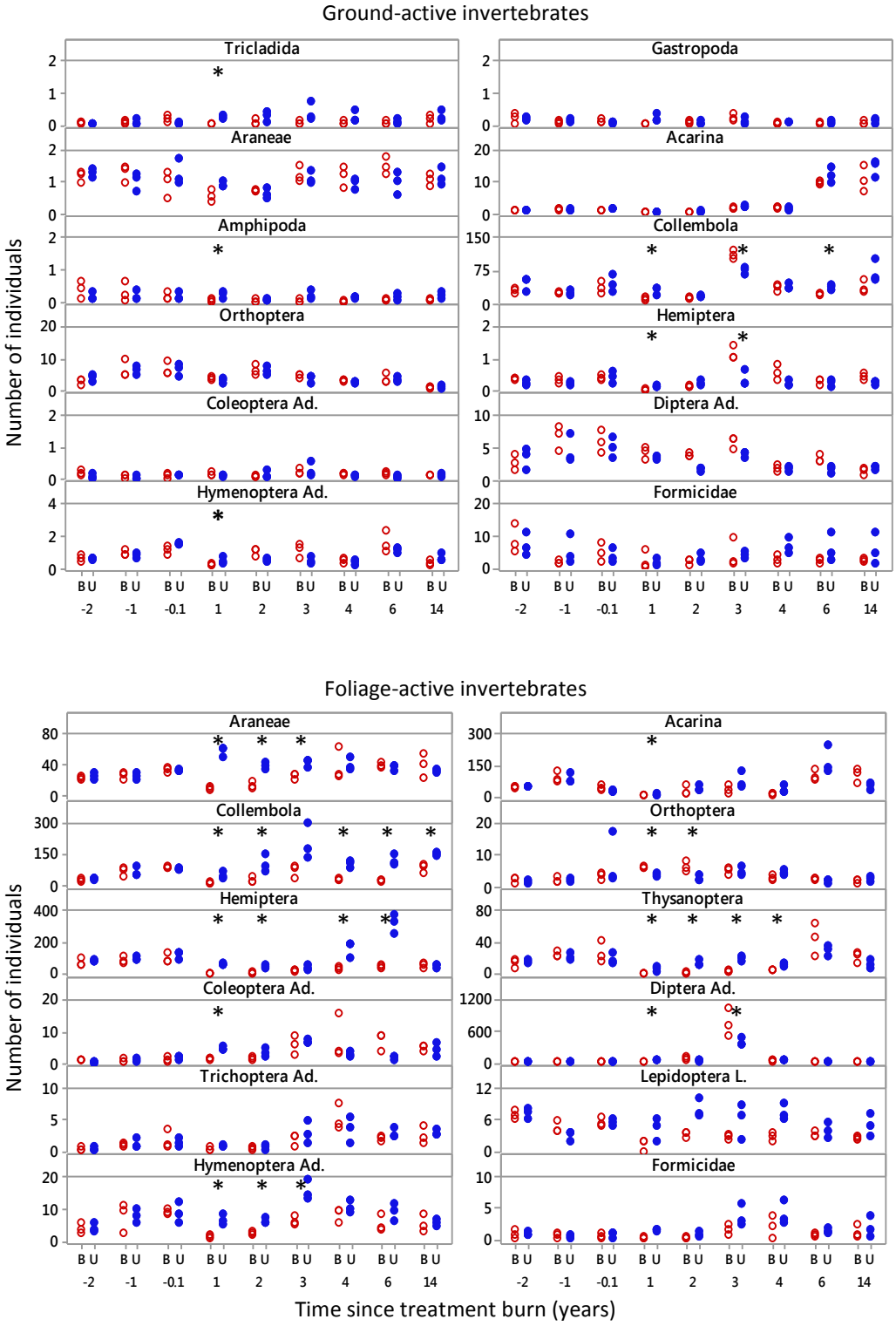

Fig. 5.3 Individual value plots of invertebrate abundance for each dominant taxonomic group at the lowland site. Each block value is based on the average of 16 pitfall traps for ground-active invertebrates and four sweep samples for foliage-active invertebrate. B, burned (red, open circles); U, unburned (blue, filled circles); *significant pairwise differences ($P < 0.05$) between treatments for global tests with significant ($P < 0.1$) time since treatment burn and treatment interaction.

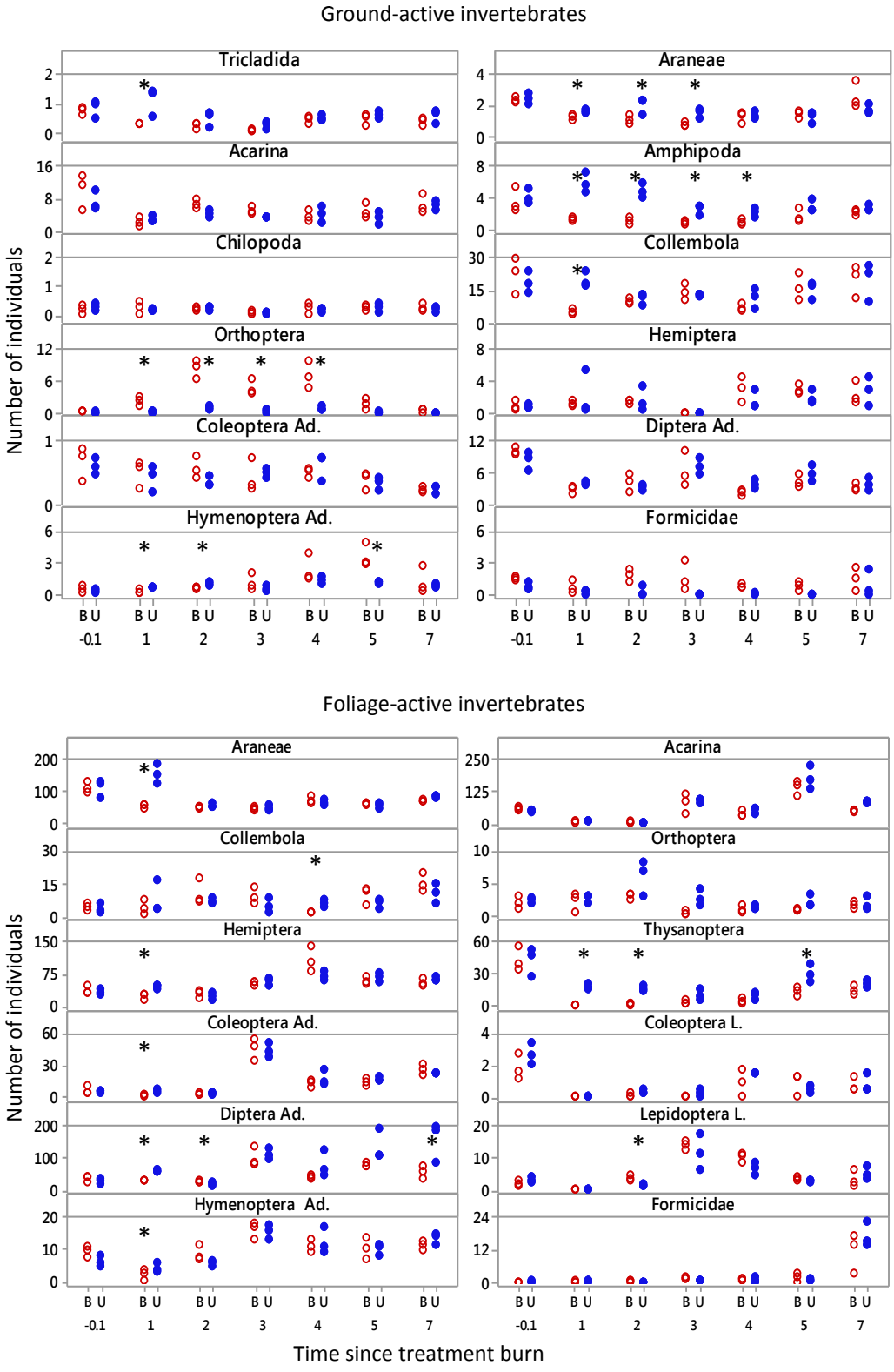

Fig. 5.4 Individual value plots of invertebrate abundance for each dominant taxonomic group at the montane site. Each block value is based on the average of 16 pitfall traps for ground-active invertebrates and four sweep samples for foliage-active invertebrate. B, burned (red, open circles); U, unburned (blue, filled circles); *significant pairwise differences ($P < 0.05$) between treatments for global tests with significant ($P < 0.1$) time since treatment burn and treatment interaction.

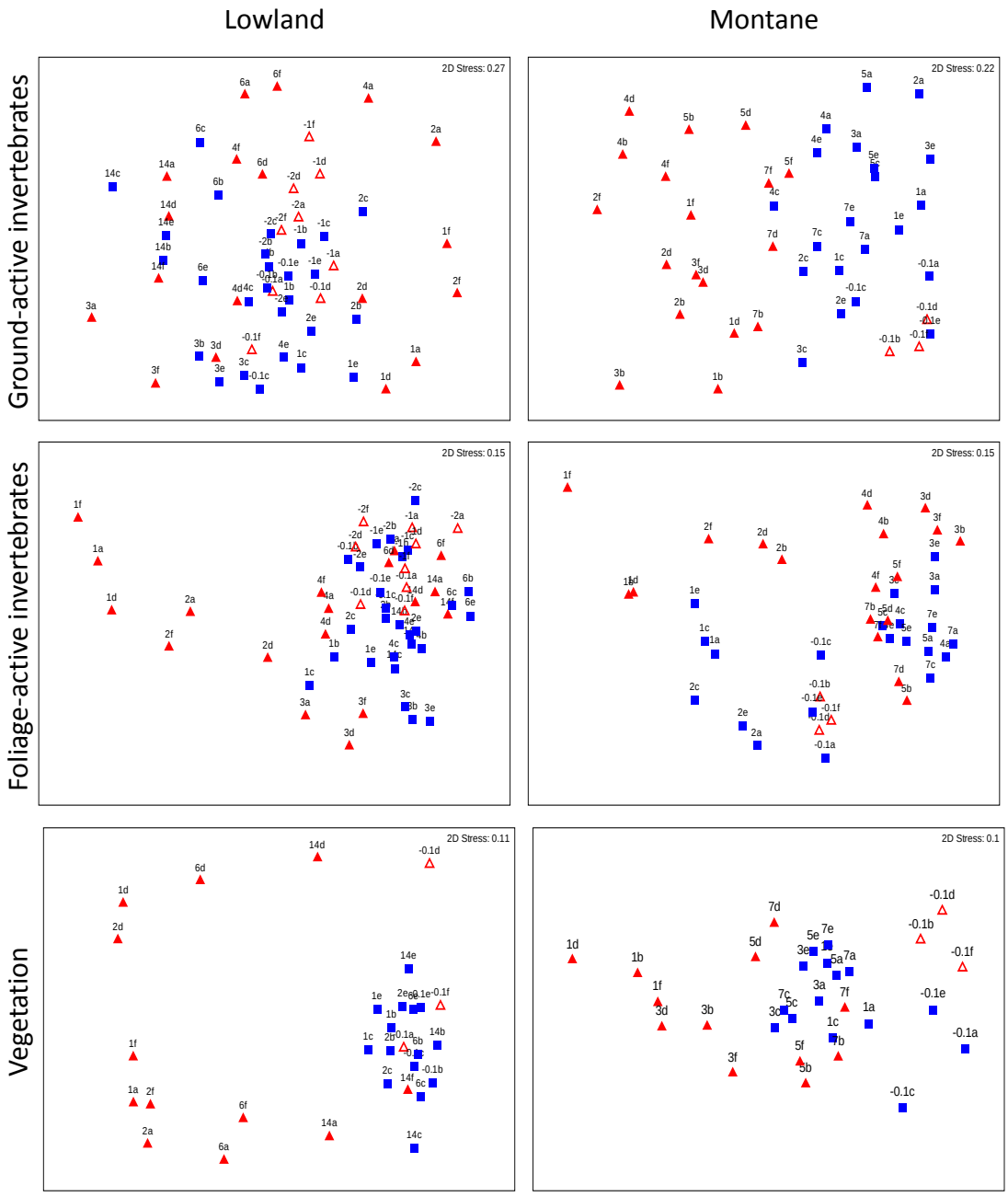

Fig. 5.5 MDS ordinations of blocks at each moorland site based on fourth root transformed ground- and foliage-active invertebrate abundances and square root transformed cover values. Numbers indicate time (years) since treatment burn and letters denote blocks. Blue squares are unburnt controls. Red triangles are treatment blocks: open triangles are pre-burn blocks and filled triangles are post-burn blocks. Note: vegetation was not surveyed every year that invertebrates were surveyed.

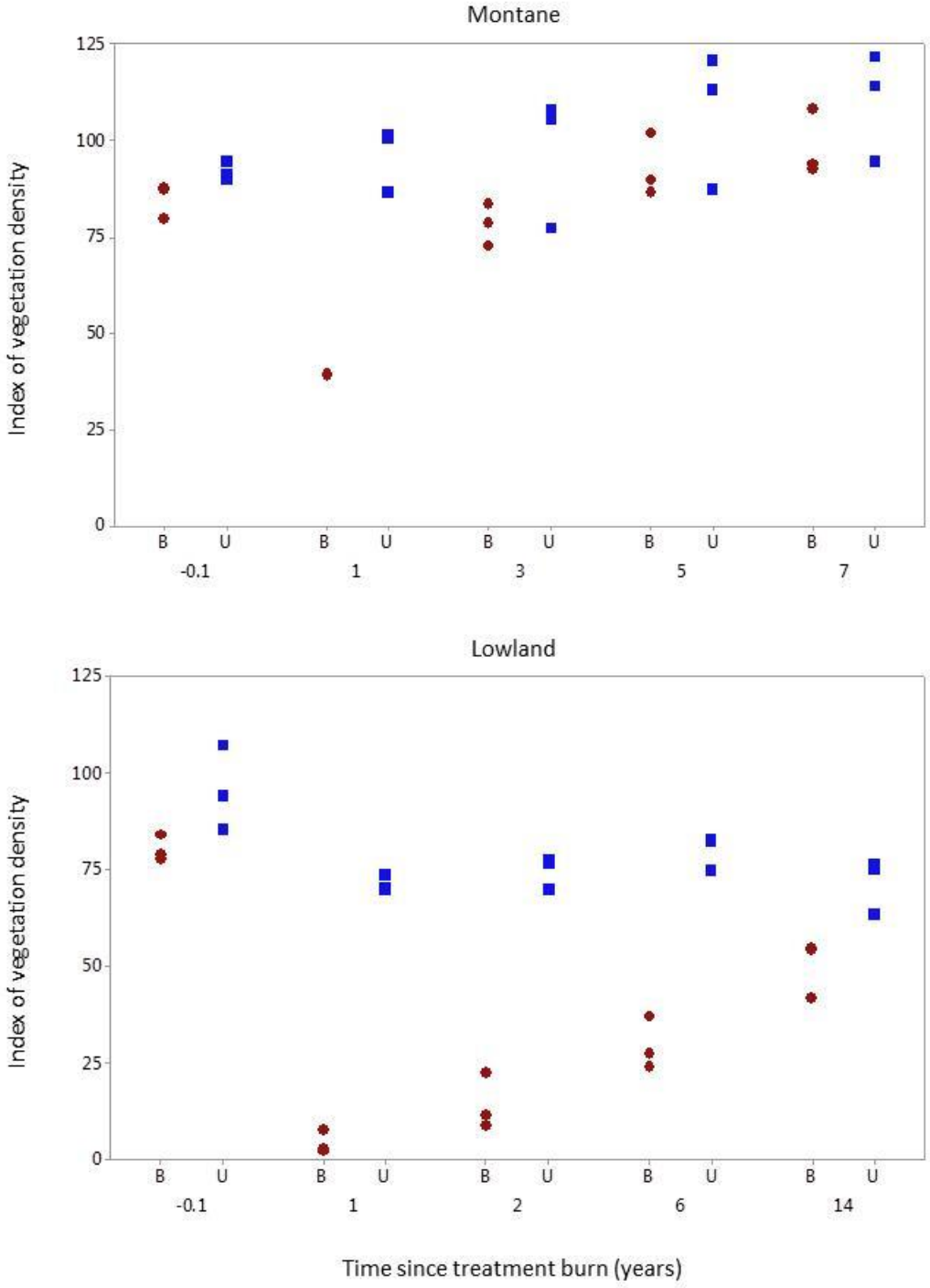

Fig. 5.6 Individual value plots of vegetation density for three burned blocks (red circles) and three unburned blocks (blue squares) for each time since treatment burn at the lowland and montane sites. Each block value is based on the average of 8 sample points. B, burned; U, unburned. Note that the time scales differ between lowland and montane sites.

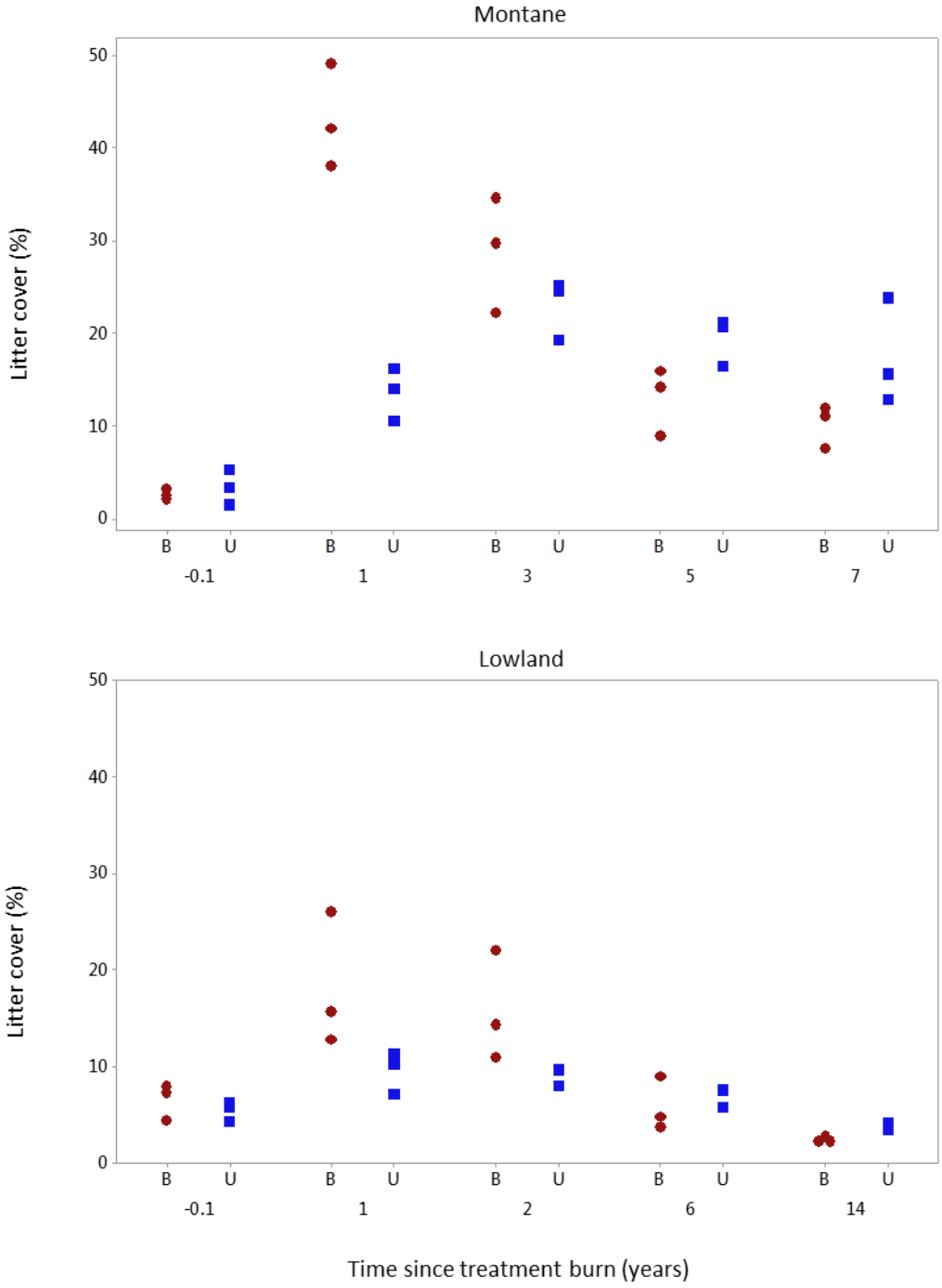

Fig. 5.7 Individual value plots of litter cover for three burned blocks (red circles) and three unburned blocks (blue squares) for each time since treatment burn at the lowland and montane sites. Each block value is based on the average of eight sample points. B, burned; U, unburned. Note that the time scales differ between lowland and montane sites.

Table 5.2 PERMANOVA based on the Bray-Curtis dissimilarities on fourth root transformed abundance data, examining effects of treatment burns on ground- and foliage-active invertebrate assemblages for lowland and montane moorlands. Each test was done using 4,999 permutations. *P* values were obtained using permutations. Square root of the component of variation attributable to that factor in the model, in units of Bray–Curtis dissimilarity. TSTB, time since treatment burn (years).

Source	df	SS	MS	Pseudo-F	*P* (perm)	Sq. root
Lowland						
Ground-active						
Burn	1	1133.00	1133.00	6.755	0.0060	6.780
TSTB	6	2736.06	456.01	5.675	0.0002	7.913
Burn X TSTB	6	1006.38	167.73	2.087	0.0002	5.397
Residual	28	2250.05	80.36			8.964
Foliage-active						
Burn	1	379.74	379.74	3.486	0.0293	3.591
TSTB	6	5803.08	967.18	21.631	0.0001	12.399
Burn X TSTB	6	653.52	108.92	2.436	0.0018	4.626
Residual	28	1251.96	44.71			6.687
Montane						
Ground-active						
Burn	1	431.85	431.85	2.313	0.0546	3.013
TSTB	8	5873.5	734.18	5.258	0.0002	9.954
Burn X TSTB	8	1493.6	186.71	1.337	0.0766	3.961
Residual	36	5026.6	139.63			11.816
Foliage-active						
Burn	1	981.71	981.71	3.907	0.0436	5.201
TSTB	8	6214.24	776.78	15.399	0.0002	11.003
Burn X TSTB	8	2010.16	251.27	4.981	0.0002	8.182
Residual	36	1815.84	50.44			7.102

Table 5.3 Multivariate pairwise comparisons of invertebrate assemblages among the two treatments (burned, unburned) at each time since treatment burn (years) using PERMANOVA and ANOSIM based on Bray–Curtis dissimilarities on fourth root transformed abundance data. PERMANOVA comparisons were undertaken following a significant Burn X TSTB interaction term in Table 5.2. The *P*-value for each test was obtained using a Monte Carlo draw from the asymptotic permutation distribution. No corrections for multiple tests have been applied.

Time since treatment burn	Ground-active invertebrates burned vs. unburned blocks			Foliage-active invertebrates burned vs. unburned blocks		
	PERMANOVA		ANOSIM	PERMANOVA		ANOSIM
	t	*P*	*R*	*t*	*P*	*R*
Lowland						
-2	0.881	0.533	-0.07	0.699	0.721	-0.07
-1	1.010	0.410	0.04	0.502	0.865	-0.22
-0.1	1.070	0.380	0.11	0.900	0.507	-0.30
1	1.640	0.089	0.44	3.914	0.003	1.00
2	1.370	0.180	0.33	3.180	0.011	1.00
3	1.340	0.181	0.26	2.375	0.021	0.89
4	1.160	0.289	0.11	2.467	0.016	0.93
6	1.320	0.198	0.26	3.798	0.003	1.00
14	1.040	0.400	-0.11	1.914	0.052	0.70
Montane						
-0.1	0.996	0.442	0.00	0.849	0.540	-0.185
1	2.112	0.029	1.00	2.806	0.022	0.852
2	2.280	0.025	0.96	2.046	0.033	0.852
3	1.965	0.045	0.74	1.398	0.160	0.519
4	2.285	0.019	1.00	1.993	0.037	0.741
5	2.099	0.042	0.70	1.095	0.349	0.296
7	1.361	0.186	0.19	1.370	0.170	0.481

Table 5.4 PERMANOVA based on the Bray–Curtis dissimilarities on square root transformed percentage cover data, examining effects of treatment burns on vegetation cover for lowland and montane moorlands. Each test was done using 4,999 permutations. Square root of the component of variation attributable to that factor in the model, in units of Bray–Curtis dissimilarity. TSTB, time since treatment burn (years).

Source	df	SS	MS	Pseudo-F	$P_{(perm)}$	Sq. root
Lowland						
Burn	1	774.07	774.07	6.278	0.0562	6.587
TSTB	4	866.39	216.60	4.821	0.0002	5.349
Burn X TSTB	4	493.19	123.30	2.744	0.0042	5.111
Residual	20	898.54	44.93			6.703
Total	29	3032.19				
Montane						
Burn	1	298.44	298.44	2.206	0.1642	3.298
TSTB	4	1135.28	283.82	10.952	0.0002	6.556
Burn X TSTB	4	541.12	135.28	5.220	0.0002	6.038
Residual	20	518.20	25.91			5.091
Total	29	2493.04				

Table 5.5 Multivariate pairwise comparisons of vegetation composition among the two treatments (burned, unburned) at each time since treatment burn (years) using PERMANOVA and ANOSIM based on Bray–Curtis dissimilarities on fourth root transformed cover data. PERMANOVA comparisons were undertaken following the significant Burn X TSTB interaction term in Table 5.4. The *P*-value for each test was obtained using a Monte Carlo draw from the asymptotic permutation distribution. No corrections for multiple tests have been applied.

Time since treatment burn	PERMANOVA		ANOSIM
	t	*P*	*R*
Lowland			
-0.1	1.075	0.351	-0.148
1	3.144	0.007	1.000
2	3.260	0.008	1.000
6	2.377	0.024	0.778
14	1.218	0.268	0.185
Montane			
-0.1	1.764	0.080	0.556
1	4.263	0.004	1.000
3	2.351	0.027	0.852
5	1.946	0.052	0.815
7	1.348	0.188	0.148

Table 5.6 The contribution (R^2) of environmental predictor variables to explaining the variation in lowland invertebrate assemblage composition when fitted alone using the DISTLM marginal test and when fitted altogether using the DISTLM multi-model approach (global model). Best model is the lowest AIC_C ranked model from the multi-model approach. Multi-model summed weight ($\sum\omega_i$) is the relative importance of each variable across all DISTLM models. *P < 0.05 from marginal test (when variable fitted alone).

Variable	Ground-active		Foliage-active	
	R^2	$\sum\omega_i$	R^2	$\sum\omega_i$
Buttongrass	0.11*	0.44	0.13*	0.50
Ferns	0.03	0.26	0.06	0.33
Herbs	0.13*	0.42	0.16*	0.39
Shrubs	0.04	0.45	0.21*	0.36
Sedges	0.07*	0.31	0.32*	0.47
Cryptograms	0.09*	0.52	0.03	0.31
Sqrt litter	0.17*	0.74	0.36*	0.81
Vegetation density[1]	0.13*	0.42	0.41*	0.66
Global model	0.48		0.68	
Best model				
Herbs + shrubs + cryptograms + sqrt litter	0.39			
Sqrt litter + vegetation density			0.51	

[1]Correlated ($r \geq |0.9|$) with cover of total vegetation, bare ground, regrowth age and vegetation height.

Table 5.7 The contribution (R^2) of environmental predictor variables to explaining the variation in montane invertebrate assemblage composition when fitted alone using the DISTLM marginal test and when fitted altogether using the DISTLM multi-model approach (Global model). Best model is the lowest AIC_C ranked model from the multi-model approach. Multi-model summed weight ($\sum\omega_i$) is the relative importance of each variable across all DISTLM models. *P < 0.05 from marginal test (when variable fitted alone).

Variable	Ground-active		Foliage-active	
	R^2	$\sum\omega_i$	R^2	$\sum\omega_i$
Sqrt bare ground	0.20*	0.57	0.14*	0.34
Grasses	0.16*	0.56	0.12*	0.51
Ferns	0.05	0.30	0.04	0.27
Sqrt herbs	0.17*	0.48	0.12*	0.46
Shrubs	0.07*	0.25	0.07	0.29
Sedges[1]	0.02	0.25	0.03	0.27
Cryptograms	0.04	0.30	0.01	0.30
Litter	0.14*	0.53	0.20*	0.64
Vegetation density[2]	0.14*	0.57	0.29*	0.93
Regrowth age	0.17*	0.44	0.13*	0.33
Global model	0.58		0.63	
Best model				
Bare ground + litter + vegetation density	0.38			
Grass + litter + vegetation density			0.52	

[1]Correlated ($r \geq 0.8$) with vegetation height and cover of buttongrass

[2]Correlated ($r = 0.8$) with cover of total vegetation

Table 5.8 The best matches between the among-sample patterns for each of four invertebrate assemblage patterns and that from environmental variables associated with those samples using BIOENV. The best one and two variable matches are also shown if they are not the overall best match. ρ is the spearman correlation coefficient. Statistical significance of top ranked matches was 0.0002 (4,999 permutations).

Ground-active		Foliage-active	
Best match	ρ	Best match	ρ
Lowland moorland			
Herb + cryptogram + litter + vegetation density	0.50	Litter + vegetation density	0.71
Litter + vegetation density	0.46	Vegetation density	0.66
Vegetation density	0.40		
Montane moorland			
Bare ground + vegetation density	0.62	Bare ground + litter + vegetation density	0.59
Bare ground	0.53	Litter + vegetation density	0.57
Vegetation density	0.37	Litter	0.57
		Vegetation density	0.42

5.5 Discussion

Using higher taxa as surrogates for species-level identification, the composition of ground- and foliage-active invertebrate assemblages shifted in response to low-intensity fire in moorlands. The abundance of most invertebrate groups was less after fire than before, which is consistent with data from elsewhere (Swengel 2001; Whelan et al. 2002). Orthoptera increased in abundance following fire, as has been found in other studies (Friend and Williams 1996; Bess et al. 2002; Hochkirch and Adorf 2007; Chapter 3), but this increase was not consistent across sites or survey methods. As expected in fire-prone ecosystems (e.g. Friend and Williams 1996; Bess et al. 2002; Brennan et al. 2006; Underwood and Quinn 2010; Pyrke and Samways 2012b; Chapter 3), the post-fire changes in invertebrate assemblage composition were correlated with post-fire changes in the composition of vegetation and other environmental elements. The strength and duration of these changes varied depending on the productivity of the moorland and whether the invertebrates were ground- or foliage- active. As was found in the space-for-time study (SFT) in Chapter 3, the shift in foliage-active invertebrate assemblage composition was greater and lasted longer at the lowland site on low productivity soil. Contrary to expectations and the results of the SFT study, this trend was not apparent for ground-active assemblages. There was only a weak short-term effect of fire on lowland ground-active invertebrate composition. In the SFT study, lowland ground-active invertebrate assemblages in three-year-old regrowth were strongly differentiated from assemblages in older regrowth (25-57 years). Although the cause of the difference between the two studies is not known, it may relate to differences in experimental design, taxonomic resolution and, in particular, site specific characteristics of the burn and ground-active fauna at the lowland site in the present study. The scale of the burns in the present study particularly at the lowland site (30 by 30 m blocks) probably meant that the burns were not as intense as in the SFT study and that this resulted in large amounts of thatch remaining two–three years post-fire. More ground-active invertebrates may have survived the less intense burns and the thatch would have provided sheltered, moist habitat post-fire. There have been previous studies in fire-prone vegetation that have also reported little effect of low–moderate intensity fire on ground-active invertebrate

assemblages (Abbott et al. 1984; Majer 1984; Coleman and Rieske 2006). The close proximity (< 20 m) of the treatment blocks to the control blocks may also have contributed to a quicker return to the pre-fire state for ground-active invertebrates.

The responses to fire for individual invertebrate groups in the present study are generally consistent with the responses reported in the SFT study (Chapter 3). The major exceptions were Formicidae and Collembola. Formicidae did not increase in abundance in recently burned moorlands in the present study as was found in the SFT study or in many other studies involving this taxon in other fire-prone communities (O'Dowd and Gill 1984a; Bess et al. 2002; Niwa and Peck 2002; Underwood and Quinn 2010; Matsuda et al. 2011). In the present study, Collembola abundance at the lowland site was lower than pre-fire 1–14 years after fire, whereas in in the SFT study, Collembola abundance at lowland sites was found to be much greater in young regrowth (3 yrs) than in old regrowth (13–57 years). The reasons for these discrepancies are not known. The small number of replicate experimental blocks (n = 3) meant that small magnitude differences between treatment effects and controls may not have been detected for some taxonomic groups.

Environmental predictors

Plant community composition and plant structural diversity are important predictors of variation in invertebrate abundance and species richness (e.g. Southwood et al. 1979; Lawton 1983; Siemann et al. 1998; Knops et al. 1999; Walter and Proctor 1999; Koricheva et al. 2000; Richardson and Hanks 2009). Thus, as expected, the present study found that shifts in invertebrate assemblage compositions following fire in buttongrass moorlands were correlated with shifts in vegetation composition. This was also demonstrated in the SFT study (Chapter 3). Also consistent with the SFT study, the present study found that vegetation density was an important predictor of variation in invertebrate assemblage composition for ground and foliage-active invertebrates at both moorland sites. Vegetation density was also correlated with other predictor variables that were affected by fire such as vegetation height, cover of bare ground and cover of total vegetation.

In contrast to the SFT study, the present study found that the cover of litter was an important predictor of variation in ground- and foliage-active invertebrate assemblage composition, especially at the lowland site. Litter has been found to be an important in predicting invertebrate assemblages (York 2000; Parr et al. 2004; Brennan et al. 2006); however, in these previous studies litter refers to detached plant material which is generally accumulated over time following fire but is largely consumed immediately following fire. Buttongrass moorlands retain dead material on the plant, which limits the amount of loose litter available (Bridle et al. 2003). However, following cool fires there may be a large amount of unburnt dead plant material forming a dense ground cover of thatch (Bridle et al. 2003; Balmer and Storey 2010). In the present study, at both sites, thatch increased after fire and persisted for at least two years before decomposing which is consistent with other observations of thatch decay in moorlands (Balmer and Storey 2010). The benefit of thatch to foliage-active invertebrates in buttongrass moorlands is not immediately apparent and it may be a proxy for another variable. However, the presence of thatch may explain the limited effect of fire on ground-active invertebrates particularly at the lowland sites by providing habitat and retaining soil moisture and reduced heating during summer months following fire. The stronger effect of fire on ground-active invertebrate assemblage composition in young regrowth in the SFT study compared with the present BACI study may due to the absence of thatch after fire because the fires were on a larger scale and would have burned hotter than the small block burns in the BACI study as discussed above.

5.6 Conclusions

Using higher taxa as surrogates for species-level identification, low-intensity fires was found to have a short-term (1–6 years) impact on ground- and foliage-active-invertebrate assemblages in buttongrass moorlands, with the abundance of most taxonomic groups less after fire and the Orthoptera more after fire. There was a longer-term effect of fire on the composition of foliage-active invertebrates in lowland moorlands that remained detectable 14 years after fire when the study concluded. There was no evidence of any invertebrate groups failing to recolonise

following fire. Vegetation density and cover of thatch were important predictors of invertebrate assemblage composition.

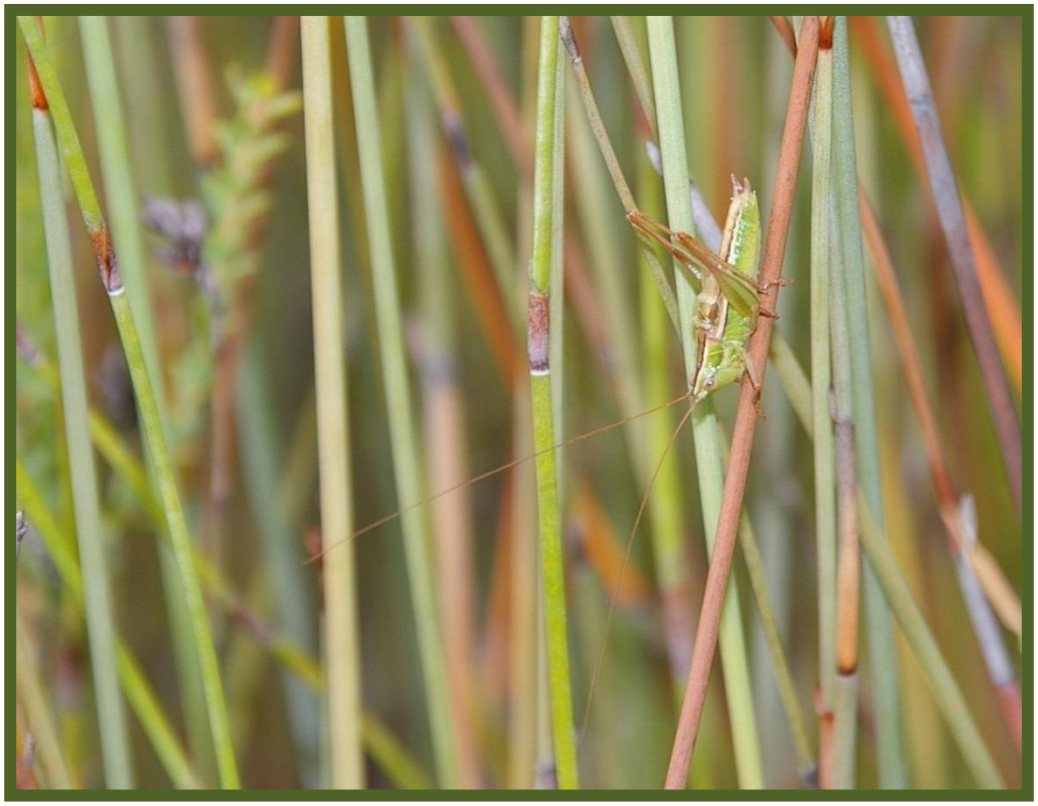

Plate 22 Meadow katydid *Conocephalus bilineatus* (Erichson) (Orthoptera: Tettigoniidae)

Plate 23 Slender ringtail damselfly *Austrolestes analis* (Rambur) (Odonata: Lestidae)

Chapter 6

Are functional traits important predictors of fire response for a broad range of invertebrate groups?

6.1 Abstract

Identification of traits that render species susceptible to fire is important for fire management planning and biodiversity conservation. Using a broad range of moorland invertebrate taxa (162 species, 21 orders), I tested whether several traits relating to dispersal, behaviour and resource use predicted responses to fire. Decaying vegetation association, adult position relative to ground, development type and juvenile trophic status best predicted invertebrate compositional response to fire, but only explained 19% of the variation. The small influence of the trialled species traits on fire response may be because some traits were not directly comparable across diverse taxonomic groups and also because of uncertainty in assigning some traits due to limited autecological data. This lack of autecological data for invertebrates is long-standing and limits the development of a broadly applied mechanistic framework for invertebrate responses to disturbance.

Key words: decaying vegetation, development type, trophic status, position in environment, Tasmania, moorland.

6.2 Introduction

Classifications based on species traits have been promoted as a promising approach for understanding the mechanisms that drive invertebrate responses to environmental disturbances (Whelan et al. 2002; Ewers and Didham 2006; Moretti and Legg 2009; Aubin et al. 2013). Although trait-based classifications have been successfully used to predict the persistence and re-establishment of plants after fire (Noble and Slatyer 1980; Keith and Bradstock 1994; Pausas et al. 2006), they are less developed for invertebrates and have not been broadly accepted or applied (Gill et al. 2002; Keith et al. 2002b). The examination of functional traits across a

broad range of taxa offers the opportunity to identify key functions that drive invertebrate responses to fire. However, most disturbance studies investigating key traits for invertebrates have focussed on one or two orders (e.g. Didham et al. 1998; Driscoll and Weir 2005; Ewers and Didham 2006; Mateos et al. 2011; Aubin et al. 2013; Moranz et al. 2013; Winqvist et al. 2014). In a multiorder, multispecies study (six orders, 471 species) in chestnut forests of Switzerland, dispersal ability, trophic level, temperature needs, position in environment, body size and moisture needs were found to be important predictors of invertebrate fire response (Moretti and Legg 2009). They found that warmth-demanding insects, herbivores, flying carnivores and pollinators were associated with recent fires whereas arthropods that were small, less mobile, moisture-demanding, ground-active and dead wood-consumers were associated with low fire frequency and unburnt sites. In American prairies using 151 species from seven insect orders, limited dispersal ability was found to be significant predictor of fire negative response (Panzer 2002). In other disturbance studies, body size, trophic level, dispersal ability and position of the environment that a species occupies have all been found to be useful in predicting species responses, with large body size, low mobility, high trophic level and species living underground more vulnerable to disturbance (Kruess and Tscharntke 1994; Driscoll and Weir 2005; Ewers and Didham 2006; Winqvist et al. 2014).

I tested whether several, relatively easily determined traits of taxa from a broad range of invertebrate groups predicted responses to fire. Because fire can directly kill invertebrates and remove critical habitat, I expected that traits associated with poor dispersal ability, limited protection from fire and reliance on habitat components that take a long time to re-establish after fire would be associated with long unburnt sites.

6.3 Methods

Data from Chapter 3 was used for the present study, which demonstrated a clear invertebrate response to fire, with invertebrate assemblage structure in the youngest regrowth clearly distinguished from older regrowth for both ground- and foliage-active invertebrates in both lowland and montane moorland. Only taxa with a minimum sample size of 25 at either study location were subjected to analysis.

The cut-off abundance of 25 equals the number of sites at the montane location and is based on the rationale that if each common taxa were distributed at random there would be an equal probability of detecting species presence at all 25 sites (Didham et al. 1998; Driscoll and Weir 2005).

A fire response index was derived for each taxon. It was the correlation coefficient between the abundance of each taxon and the age of regrowth axis obtained from a canonical analysis of principal coordinates performed on a matrix comprising all taxa, all sites and two sampling methods, and using age of regrowth to discriminate among the invertebrate assemblages. The fire response index varied from –0.39 (taxa associated with old regrowth) to 0.41 (taxa associated with young regrowth); median = –0.06.

Six classes of functional traits were used: size (minute, <1 mm; small, 1–5 mm; medium, 6–10 mm; large, >10 mm); development type (complete metamorphosis, no/gradual metamorphosis); flight (flying, flightless); adult and juvenile trophic level (phytophage, carnivore, parasite, saprophage—including fungivores, coprophages and microbivores); adult and juvenile primary position (underground/on ground, aboveground, aquatic); and an association with decaying vegetation (yes, no) reported in published literature.

DISTLM was used to explore and model the relationship between the fire response index and species traits. Traits were included as sets of categorical predictor variables as described in Anderson et al. (2008). Marginal tests were performed to assess the statistical significance and percentage contribution of each trait alone. Models were constructed from all possible combinations of the six traits using the BEST selection procedure and the corrected Akaike Information Criteria (AICc) were used to rank the models. The model with the lowest AIC_c value ($AIC_{c(min)}$) is the estimated 'best' of the candidate models. Because models within seven AIC_c units of $AIC_{c(min)}$ have some support and should rarely be dismissed (Burnham et al. 2011), the relative importance of predictor variables were assessed and ranked by summing AIC_c model weights across all models that included that variable (Burnham and Anderson 2002). The summed AIC_c weights for each variable can be interpreted as equivalent to the probability that the variable is a component of the best model (Symonds and Moussalli 2011).

6.4 Results

Only a small proportion (23%) of the variation in fire response index could be explained by all eight species traits (Table 6.1). Adult position, juvenile position and association with decaying vegetation explained significant but negligible amounts of variation when fitted alone. The model containing four variables—association with decaying vegetation, juvenile trophic status, adult position and development type–was the estimated 'best' model explaining 19% of the variation, and these four variables received strong support from summed AIC_c weights. Taxa that occurred above ground as adults, that were associated with decaying vegetation, that were juvenile parasites and that had gradual or no metamorphosis tended to be associated with old regrowth moorland (Fig. 6.1).

6.5 Discussion

Previous studies have found trophic status, adult position and association with decaying vegetation (or moisture requirement) to be predictors of responses to disturbance (Driscoll and Weir 2005; Moretti and Legg 2009; Mateos et al. 2011; Aubin et al. 2013) but I am not aware of studies that have considered development type. However, Swengel (1996) recognised that above-ground stages of butterflies were more susceptible to fire than stages that sheltered underground or in other refuges. Unlike earlier studies (Panzer 2002; Aubin et al. 2013), there was little evidence that body size and ability to fly were important traits predicting response to fire.

Most studies have not assessed and discussed the amount of variation in disturbance response explained by species traits. The small influence of species traits to predict responses to fire in our study may relate to uncertainty in applying some traits, such as species vagility and trophic status, to some groups. Also, some traits may be good predictors of fire response for particular invertebrate groups and not for others and this information may have been lost by conducting the analysis across a broad range of invertebrate groups. For example body size has been found to be a predictor of disturbance response in Coleoptera (see review by Ewers and Didham 2006), but comparisons of Coleoptera body size with Acarina or Collembola

in the present study are crude. Scaling body size within each group may provide better insights. Similarly, ability to fly or not may also be a crude indicator of dispersal ability when considering a broad range of invertebrate groups, as many non-flying taxa are able to disperse significant distances (at least in the context of the present study) with the assistance of wind and water (e.g. Araneae, Acarina and Collembola). Thus, some traits may not be directly comparable across diverse taxonomic groups and therefore confound the usefulness of the traits investigated. Use of other traits that could not be readily applied due to lack of autecology data (e.g. life cycle duration, number of generations per year) may improve the predictive ability of species traits across a broad range of taxa.

The need for a mechanistic understanding of fauna species responses to fire regimes is long-standing (Whelan 1995; Driscoll et al. 2010) and highlights the difficulties in autecology data, particularly for invertebrates.

6.6 Conclusion

Using a broad range of moorland invertebrate taxa (162 species, 21 orders), trophic status, position in the environment, association with decaying vegetation and development type best predicted invertebrate compositional response to fire, but only explained 19% of the variation. The functional traits used in this study require further investigation before they can be considered useful in predicting the response of invertebrate taxa to fire and other traits need to be considered.

Table 6.1 The contribution (R^2) of each functional trait to explaining the variation in fire response index and their relative importance when their AIC_c weights ($\sum\omega_i$) are summed across all DISTLM models in the multi-model approach. Highest ranked variables are in bold. Global model is the contribution of all eight variables to explaining the variation in fire response index. *P < 0.05 from marginal test (variables fitted alone)

Functional trait	R^2	$\sum \omega_i$
Size class	0.01	0.12
Development type (DT)	0.001	**0.77**
Flight	0.000	0.36
Adult trophic status	0.002	0.15
Juvenile trophic status (JT)	0.03	**0.87**
Adult position (AP)	0.06*	**0.73**
Juvenile position	0.05*	0.43
Decaying vegetation association (Decay)	0.03*	**0.99**
Best model		
DT + JT + AP + Decay	0.19	
Global model	0.23	

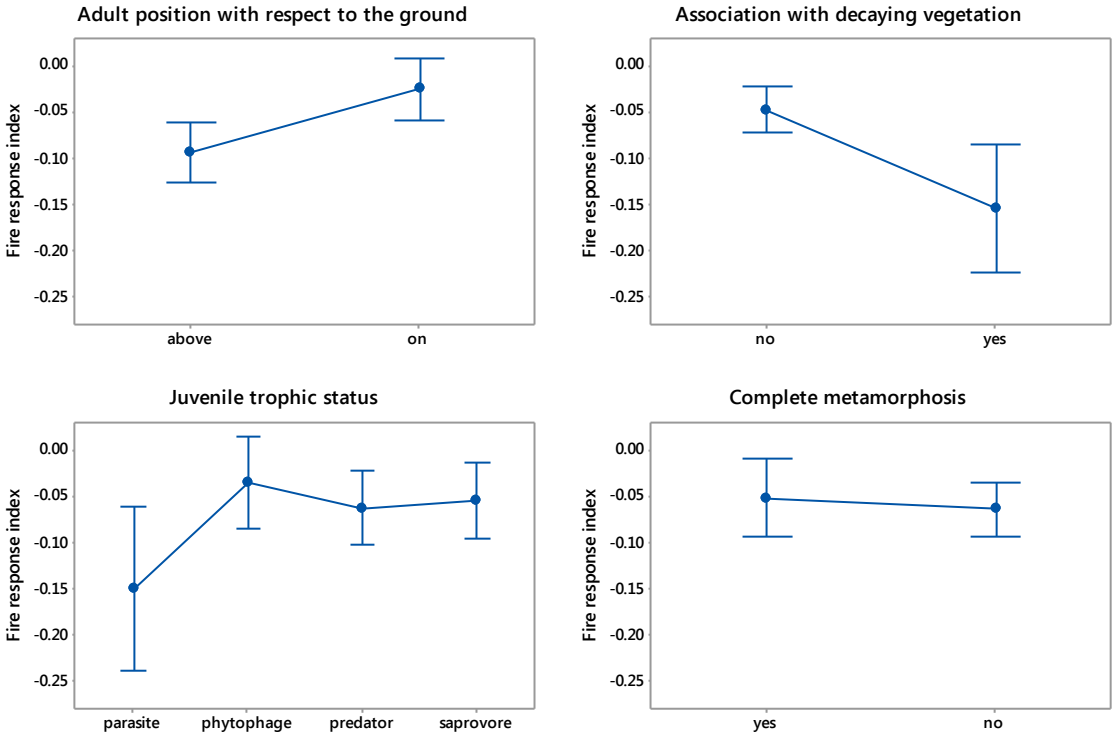

Fig. 6.1 Relationships between the fire response index and four species traits that in combination were the best predictors of variation in the fire response index. Larger negative values of fire response index indicate greater association with older regrowth. 95% confidence intervals for the mean are shown. The pooled standard deviation was used to calculate the intervals

Plate 24 Lappet moth caterpillar *Pterolocera* sp. (Lepidoptera: Anthelidae)

Plate 25 Two-lined Euloxia *Euloxia meandraria*. (Guenée) (Lepidoptera: Anthelidae)

Chapter 7

General Discussion

7.1 Introduction

The overall objective of this thesis was to investigate the resilience of moorland invertebrate assemblages to low-intensity fire and to identify predictors of assemblage change and its return to the pre-fire state. My approach differed from previous studies in that:

- I used two complementary research designs: (1) a before-after-control impact (BACI) design that aimed to minimise spatial variation, and (2) a replicated space-for-time (SFT) design that provided insight into longer-term invertebrate responses to fire, and had a broader spatial scale, than the BACI design.
- Both research designs had a long-term post-fire monitoring focus aimed at determining if and when ground- and foliage-active invertebrate assemblages returned to the pre-fire state.
- Both research designs used a wide-range of invertebrate groups, with species-level identification being used for the SFT study.
- The effectiveness of using higher taxa as surrogates for species-level identification to assess invertebrate assemblage fire response was tested.
- Seasonal variation in species diversity and abundance was investigated to identify the optimal time to survey invertebrates and the limitations associated with surveying at particular times.

In Chapter 1, I presented five specific questions that would be addressed by this thesis; they are:

1 Are moorland invertebrates resilient to fire?

2 Do moorland invertebrate assemblages change following fire and is there a pattern of succession?

3 Are there moorland taxa that are sensitive to fire?

4 Are functional traits important predictors of invertebrate response to fire?

5 Are higher taxa effective surrogates for species-level identification in representing patterns of assemblages change due to fire?

The aim of this chapter is to synthesise the results of chapters 2, 3 and 5 in relation to questions 1–3, which primarily relate to the overall objective of this thesis. To avoid repetition, questions 4 and 5 are addressed briefly below because chapters 4 and 6 focused specifically on each of these questions respectively. I then discuss the implications of my research, from all chapters, for future fire management and research.

In relation to question 4, I found that the species traits tested in the present study were poor predictors of invertebrate response to fire. The small influence of species traits on responses to fire observed in the present study may relate to the diverse range of taxa used in my study, with some traits not directly comparable across groups.

In relation to question 5, as expected, higher level taxa were found to be effective surrogates for species-level identification in representing patterns in invertebrate assemblage structure and detecting effects of a major disturbance event (fire), with data aggregated to family level being more effective than data aggregated to order level.

7.2 Are moorland invertebrates resilient to fire?

Buttongrass moorland is a fire-prone vegetation type that was burnt by hunter-gathers over thousands of years to facilitate easy passage and to encourage game (Thomas 1993). It has been inferred from accounts by early settlers and the close proximity of fire sensitive vegetation to moorland that indigenous people conducted frequent, low-intensity fires (Marsden-Smedley and Kirkpatrick 2000). Thus, it is reasonable to expect that the invertebrate community of buttongrass moorlands has evolved to be resilient to fire, with any fire sensitive taxa lost from the community long ago or restricted to fire refugia where moorland is the climax

community. Consistent with this expectation, fire impacts on ground- and foliage-active invertebrate assemblages in moorland were limited to short–medium term changes in abundance, with no medium–long-term loss of species or higher taxa (Chapters 2, 3 and 5). Thus the sampled components of the invertebrate fauna appear resilient to at least single low-intensity fires (Chapters 3 and 5). They also appear to be resilient to a regime that over 71 years comprised multiple fires (with a mean fire interval of 24 years in lowland moorlands and 37 years in montane moorlands) including high-intensity summer fires (Chapter 3). This resilience of invertebrates to fire has been observed elsewhere in the world particularly in fire-prone vegetation types (Friend and Williams 1996; Andersen and Müller 2000; Bess et al. 2002; Moretti et al. 2004; Parr et al. 2004; Brennan et al. 2006; Pyrke and Samways 2012a; New 2014).

An important caveat on the resilience of moorland invertebrates to fire is that only ground-and foliage-active invertebrates were surveyed in the present study and that pitfall traps and sweep nets sample an unknown proportion of these assemblages. For example, nocturnal foliage-active invertebrates are not sampled by daytime sweep sampling. Most species that were sampled were represented by less than 10 specimens (Chapters 2 and 3). Further, because of logistical and funding constraints, sampling was conducted only in late summer to maximise species richness and abundance. However, this time period accounts only for a third of ground- and foliage-active taxa (Chapter 2). It may be inferred from the present study and work elsewhere in fire-prone communities, that the non-sampled ground- and foliage-active invertebrates are also resilient to fire. However, there are indications that other invertebrate assemblages such as soil-active invertebrates may be more vulnerable to fire; or at least take longer to re-establish after fire. Planned fire and bushfire can result in substantial soil losses in Tasmanian moorlands, with greater losses on slopes than on flats (di Folco and Kirkpatrick 2011). Soil depth and in particular the depth of organic material is known to influence invertebrate assemblage composition (Mitchell 1978; Giller 1996; Marra and Edmonds 1998; Maraun and Scheu 2000). Although the present study did not survey soil-active fauna and only surveyed moorlands occurring on flats or gentle slopes, peat depth was found to have potential in predicting variation in lowland

ground-active invertebrate assemblages (Chapter 3). An investigation of soil mites using many of the same sites used in the SFT study (Chapter 3) found that both their density and diversity was significantly higher in 30–40 year old regrowth moorland compared with younger regrowth suggesting that these populations take a long time to return to high levels of density and diversity after fire (Green 2009). A complete description of the soil-active Acarina has yet to be published and further investigation of the response to fire by soil-active invertebrates is warranted. Climate change may further exacerbate the possible vulnerability of soil invertebrates to fire if predictions that peats in southwest Tasmania will become increasing flammable under the current trend of warmer and drier climates (Bowman 2008) prove to be correct.

Another important caveat on the resilience of moorland invertebrates to fire is that not all aspects of a fire regime (e.g. intensity and season) were investigated. Key attributes of a fire regime can influence the response of invertebrates to fire and these attributes are often not recorded in invertebrate studies (see reviews by Whelan et al. 2002; Parr and Chown 2003). The present study primarily investigated the effect of single, low-intensity (<500 kWm^{-1}) fires on invertebrate assemblages. However, all sites included in the present study are likely to have been burnt periodically over thousands of years. Most had also been subject to high-intensity, summer wildfires at least once since European settlement in Tasmania, with major fires occurring in 1897/98, 1933/34, 1950s and 1972 (Marsden-Smedley 1998; Johnson and Marsden-Smedley 2001), and more recently the sites have been subject to at least one low-intensity planned burn (Chapter 3). The limited research on different fire regimes and management implications are discussed further in section 7.7.

7.3 Do moorland invertebrate assemblages change following fire and is there a pattern of succession?

Fire in moorland altered plant community composition, decreased vegetation density and increased the cover of bare ground (Chapters 3 and 5). Because plant community composition and plant structural diversity are well-recognised predictors of variation in invertebrate abundance, species richness and composition in successional processes (e.g. Southwood et al. 1979; Lawton 1983; Siemann et al.

1998; Knops et al. 1999; Walter and Proctor 1999; Koricheva et al. 2000; Schaffers et al. 2008; Richardson and Hanks 2009), it is hardly surprising that fire in moorland changed the assemblage composition of ground- and foliage-active invertebrates, and this effect is consistent with numerous invertebrate fire studies (see reviews by Warren et al. 1987; Friend 1995b; Whelan 1995; Swengel 2001; Whelan et al. 2002; New 2014). However, to develop effective management plans and to develop theory for species responses to fire, it is important to define the bounds of species assemblage change after fire (Driscoll et al. 2012).

Successional patterns with time-since-fire can show a continuum of responses from a single, deterministic trajectory and endpoint to multiple trajectories and endpoints (including stable or stochastic endpoints) (Driscoll et al. 2010; Langlands et al. 2012). The deterministic habitat accommodation model describes the orderly succession of species following disturbance, with species entering and leaving the succession according to their habitat and competitive requirements (Fox 1982). The model is useful for describing sequences of species succession after fire for some small mammal, bird and reptile assemblages (Fox 1982; McFarland 1988; Friend 1993; Taylor and Fox 2001; Letnic et al. 2004). However, successional patterns for other vertebrate assemblages appear more stochastic and do not show predictable post-fire succession. They correlate poorly with structural variables indicative of vegetation succession or habitat type (Masters 1993; Letnic et al. 2004). The habitat accommodation model is likely to be useful for invertebrates because they show a strong response to habitat composition and structure. However, the model has been rarely considered in relation to this group of animals (Gosper et al. 2015).

In moorland, changes in composition of vegetation and ground- and foliage-active invertebrates followed a highly deterministic succession after low-intensity fire, with convergence towards the long-unburnt state (Chapters 2, 3 and 5). The invertebrate community compositions could be arrayed along a single continuum of vegetation composition change and conformed to a directional model with a single endpoint. Shrubs, sedges and decaying vegetation, which were mostly consumed by fire, increased in cover with time-since-fire whereas cover of bare ground and thatch decreased with time-since-fire (Chapters 3 and 5). Herbs and grasses

increased in cover during the early successional stages before being crowded out by the dominant vegetation groups (Chapter 3 and 5). Most moorland invertebrate taxa were less abundant immediately after fire than before, consistent with many other fire invertebrate studies (see reviews by Whelan 1995; Swengel 2001; Whelan et al. 2002) and then returned to pre-fire levels as vegetation and other habitat components re-established. Several invertebrate species increased in abundance during the early successional stages following fire, most notably the cricket *Bobilla poene* and two species of dolichoderine ant: *Anonychomyrma* ?*nitidiceps* and *Iridomyrmex* sp. 1. Increased abundance of many Orthoptera and Formicidae (notably Dominant Dolichoderinae) in early successional stages following fire has been reported in a wide range of habitats, and has typically been associated with post-fire increases in food resources in the forms of grasses, herbs and seeds (Evans 1984; O'Dowd and Gill 1984a; Bock and Bock 1991; Underwood and Christian 2009). In the SFT study (Chapter 3) increases in Orthoptera and Formicidae were also associated with increases in cover of herbs and grasses. For crickets, increases in area of bare ground following fire also provides a potential increase in both the area and temperature of ovipositing sites, which can increase the number and rate of egg and nymph development (Evans 1984). Contrary to expectations, no increase in Formicidae following fire was recorded in the BACI study (Chapter 5) despite an increase in grasses and herbs. Increases in Orthoptera, Formicidae and other ground-active taxa following fire may also be due to increases in activity due to more bare ground (Melbourne 1999; Driscoll et al. 2012).

It is implicit in the habitat accommodation model that successional changes in vegetation, not time *per se*, drive the response by animal communities (Monamy and Fox 2000; Monamy and Fox 2010). This conclusion was supported by the results of the present study (Chapters, 2, 3 and 5). For the most part, invertebrate assemblages in moorlands on low productivity soils took approximately twice as long to return to the pre-fire state than assemblages on moderate productivity soils. Only in lowland moorland in the BACI study did the ground-active invertebrate assemblage return to the pre-fire state more quickly than expected based on the vegetation composition (Chapter 5). The reason why lowland ground-active invertebrates did not closely follow the expected trend is not known but possibly

relates to site or fire-regime specific features that only affected them. The large cover of thatch up to two years after the small-scale, low-intensity fire in the BACI study may have ameliorated the effects of fire on this component of the fauna (Chapter 5). This raises the possibility of more than one trajectory towards the long-unburnt state, as has been found for spiders in an arid zone habitat (Langlands et al. 2012).

Clarke (2008) cautioned against the use of plant succession as a surrogate for animal succession, particularly without research into the relationship. Many animal species depend on structural aspects of vegetation (e.g. tree hollows, logs, litter, decaying vegetation, dense scrub). These attributes may have different timeframes for development compared to plant assemblage composition or seed bank development. Structural attributes, such as tree hollows and fallen logs can take much longer to develop than floristic quasi-stability. Even in a structurally challenged community like moorland; structure appears to be more important than plant community composition in determining the return of the invertebrate fauna to the pre-fire state. Vegetation density, vegetation height and litter were important predictors of invertebrate assemblage change and its return to the pre-fire state, especially for foliage-active invertebrates, which sometimes took longer to return to the pre-fire state than ground-active invertebrates (Chapters 3 and 5).

Broadly, the successional pattern of invertebrate assemblage response to low-intensity fire in moorland was highly deterministic, with convergence towards the long-unburnt state, and was mediated by the post-fire return of vegetation and other habitat elements to the pre-fire state. This pattern of response mirrors many other studies in fire-prone ecosystems ecosystems (Warren et al. 1987; Moretti et al. 2002; van Heurck and Abbot 2003; Moretti et al. 2004; Brennan et al. 2006; Pyrke and Samways 2012a; Kim and Jung 2013).

7.4 Are there moorland taxa that are sensitive to fire?

A generally accepted conservation objective for fire management is to avoid population extinctions within a defined area of management due to the effect of an adverse fire regime (Bradstock et al. 1995; Keith et al. 2002b; Driscoll et al. 2010; Kelly et al. 2015). Species with strong preferences for early or late successional

stages may be at heightened risk of extinction (Driscoll and Henderson 2008). Management should be directed at groups of species with ecological traits which render them most susceptible to decline with a change in fire regimes (Keith et al. 2002). In the present study, most ground- and foliage-active moorland invertebrates were present throughout the succession, with very few species absent or in very low abundance during early (≤ 3 years) successional stages (Chapter 3). Several taxa appeared to be more sensitive to the effects of fire by taking longer than three years to return to pre-fire levels of abundance. This group included species of ground-active Amphipoda and foliage-active Acarina, Collembola, Hemiptera and Thysanoptera (Chapters 2, 3 and 5). Remarkably, foliage-active Collembola at the lowland site remained lower in abundance on burned than unburned grids 14 years after the fire, even though the vegetation composition had returned to the pre-fire state (Chapter 5). Comparison of the responses to fire by these taxa with studies conducted elsewhere is fraught because of differences in methods, habitats, limited lengths of surveys and strengths of experimental designs. Consequently various responses to fire have been reported for these taxa (see reviews by Warren et al. 1987; Friend 1995b; New 2014). However, the few studies that have considered ground-active Amphipoda suggest that their abundance is reduced by fire and they may take much longer to increase in abundance following fire than most other invertebrate taxa (York 1999a; Barratt et al. 2009), probably because they are prone to desiccation and require the development of moist, humid microhabitat (Friend and Richardson 1986). Typical of most invertebrate studies (New 2014), there were large numbers of ground- and foliage-active species that were recorded in so few numbers that little can be concluded about whether or not they preferred early or late successional stages. Other moorland invertebrate assemblages, such as the soil-active and water-active, were not assessed and may yield taxa with strong preferences for particular successional stages. Some soil-active mites may have strong preferences for late successional stages (30+ years) in both low and moderate productivity moorland (Green 2009).

7.5 Management Implications

The over-riding principle for fire management in Tasmanian protected areas is that bushfire suppression takes priority over all other activities, especially if human life and property is threatened (PWS Fire Management Policy 2014). The policy also states that fire regimes that conserve biodiversity will be implemented based on best available knowledge, and that all reasonable steps will be taken to ensure that the impact of planned fires and fire suppression activities on natural values will be minimised. Fire management in Tasmania uses a zoning system that specifies different objectives for each zone and provides fire regime guidelines to meet those objectives (DPIPWE 2011). The four zones are:

- An Asset Zone which contains a human-made or natural feature of significant value that is negatively affected by fire.
- An Asset Protection Zone aimed at protecting values in Asset Zones by minimising bushfire risk and maximising bushfire suppression on a priority basis.
- A Strategic Fuel Management Zone aimed at increasing the likelihood of controlling a bushfire within or the forward spread through the zone.
- A Land Management Zone aimed maintaining appropriate fire regimes for vegetation communities, species diversity and cultural heritage.

Planned burning is conducted primarily in Asset Protection and Strategic Fuel Management zones but may also be conducted in Land Management Zones when and where there is an identified need for species or communities that require fire (DPIPWE 2011). Because buttongrass moorland is highly flammable it is predominantly zoned within Asset Protection and Strategic Fuel Management zones (DPIPWE 2011, 2012).

Table 7.1 Recommended fire regimes for maintenance of natural values in buttongrass moorland (Source: DPIPWE 2015)

Buttongrass Moorland Class	Minimum interval	Maximum interval	Season	Fire intensity	Comments
Low productivity	Strategic Fuel Management Zone: • 15 years Asset Protection Zone: • 15 years or: • 10 years only where soil cover is continuously deep. Medium confidence – all disciplines agree.	60 – 70 years. Low confidence. May not require fire at all, unless regional context means intervention is required.	No clear desirable season, exclude summer. Moderate confidence. Soils suffer from high temperature after spring burns or rain erosion after autumn burns. Flora specialists suggest diversity in absence of specific knowledge. Fauna specialists suggest restricting fire from spring breeding season – burn before 30 September (low confidence around date).	Low enough to leave thatch. High confidence.	Slopes versus flats – if practical lowest possible intensity on slopes above 7°. A mosaic of burnt and unburnt areas is required within burn blocks, and also at a regional scale. Within block patterns, leave patches as refuges, either within burn area or immediately adjacent. The greater the burn area, the more unburnt patches are required within block. Block shape may be an important control on dispersal, but more research is needed to make recommendations. Regional and block scale mosaics should be a focus of future research. Discussion with fire management to develop practical numerical guidelines if required.

Buttongrass Moorland Class	Minimum interval	Maximum interval	Season	Fire intensity	Comments
Sparse (TASVEG Code: MBR)	Strategic Fuel Management Zone: • 40 - 60 years. Asset Protection Zone: • Unknown. Low confidence.	None. Moderate confidence.	See Low productivity.	See Low productivity.	Recent evidence suggests there is a correlation between shallow soils and low vegetation biomass, suggesting soil formation rates may be slower in these areas.
Moderate productivity (e.g. MBE, MBP MBU in east)	Strategic Fuel Management Zone: 10 years. Asset Protection Zone: 6 years. Medium confidence – no geo data available.	40 years. Low confidence.	See Low productivity.	See Low productivity.	

In 2015, guidelines on appropriate fire regimes to maintain natural values in the Tasmanian Wilderness World Heritage Area (TWWHA) were developed for land managers by the Department of Primary Industries, Parks, Water and Environment (DPIPWE 2015). The TWWHA covers almost a quarter (1.58 million hectares) of Tasmania, with most (63%) buttongrass moorland occurring within this area. Guidelines were developed for each of the major vegetation types, with buttongrass moorland divided into three subtypes: low productivity, moderate productivity and sparse (Table 7.1). Below, I consider the fire regime guidelines for low and moderate productivity moorlands in relation to ground- and foliage-active invertebrates and previous research on buttongrass moorland invertebrates. The third moorland subtype, 'sparse moorland' was not investigated in the present study because it was only developed in the 2015 report. It is an extremely low productivity moorland with sparse vegetation cover and very shallow stony soils, and thought to have very low soil formation rates (DPIPWE 2015).

The results of the present study have, in part, already contributed to the development of DPIPWE's guidelines; in particular the need to separate moorlands into low and moderate productivity. The DPIPWE guidelines relate primarily to maintaining ecological values; however in Asset Protection Zones these and other values are of secondary importance and may be adversely impacted (Marsden-Smedley 2009). To a degree, this difference in objectives between Asset Protection Zones and Strategic Fire Management Zones is reflected in DPIPWE's guidelines (Table 7.1). The DPIPWE guidelines for buttongrass moorland do not mention Land Management Zones because the majority of this vegetation type occurs within the Strategic Fuel Management Zone, and although not stated, the guidelines for Strategic Fire Management Zones apply to Land Management Zones (K. Storey, DPIPWE pers. comm.).

Fire Interval

From an ecological perspective, fire interval (time in years between fires) is considered to be the most appropriate measure of fire frequency because it relates directly to time-dependent life history processes (Gill et al. 2002). The DPIPWE guidelines provide a minimum and maximum fire interval range for buttongrass

moorland and the low level of confidence in these values is acknowledged (Table 7.1) owing to the limited amount of research undertaken on the effects of fire frequency on soils, plants and animals (Balmer and Storey 2010; Driessen 2010; Storey 2010). More widely, the effect of fire frequency on invertebrate assemblages has been recognised as an under-researched predictor for biotic responses to fire management (Gill et al 2002). The present study primarily investigated the effect of single fires on invertebrate assemblages in moorlands subject to a low fire frequency (mean fire interval of 24 years for 44 sites), and the factors that predict their return to pre-fire states. This information provides some guidance on the minimum fire interval but not whether moorland invertebrates can continue to return to pre-fire states recorded in the present study if the minimum fire interval is continually applied. It is not the intention of the DPIPWE guidelines to continually apply the minimum value but to provide a fire interval range. This is consistent with Keith et al. (2002b) who recommend that fire management targets should be defined as ranges rather than optima to provide greater flexibility to deal with uncertainty (e.g., unplanned fires) and to resolve potential management conflicts (e.g., species that require different optima). The results from the SFT study in chapter 3 suggest that ground- and foliage-active invertebrates are resilient to a mean fire interval of at least 24 years in low productivity moorlands and 37 years in moderate productivity moorlands. Some of the sites used have experienced fire intervals of 19–22 years on low productivity moorlands and 7–13 years in moderate productivity moorlands with no obvious influence on the invertebrate assemblages (Fig. 3.2).

The minimum fire interval of 15 years suggested in the DPIPWE guidelines for low productivity moorlands (Table 7.1) does not generally allow for the return of ground- and foliage-active invertebrate assemblage compositions to the pre-fire states, which may take from 15 to 24 years after a single fire (Greenslade and Driessen 1999; Chapters 3, 5). However, all taxa (excluding rarely caught taxa) were present in moorlands three years post-fire albeit some in lower abundance. DPIPWE's guidelines also allow for a minimum fire interval of 10 years for planned burns in Asset Protection Zones with low productivity soils where soil cover is continuously deep (Table 7.1). This is based on the assumption that the soils in

these circumstances may be able to cope with more frequent burning. Impacts of 10-year interval burning on invertebrates in these circumstances have not been investigated.

In moderate productivity moorlands, the six year minimum fire interval suggested in the DPPWE guidelines generally allows for the re-establishment of both ground- and foliage-active invertebrates with mostly weak assemblage composition differences detected five–seven years after a single fire compared with longer unburnt regrowth, and by 10 years post-fire the assemblages had re-established.

Longer minimum fire intervals may be required for soil-active invertebrate assemblages in both low and moderate productivity moorlands. The only study of soil-active invertebrates in relation to fire found that Acarina diversity and abundance increased markedly 30 years post-fire in both low and moderate productivity moorlands (Green 2009). These data have yet to be fully analysed. Further investigation of the resilience of soil-active invertebrate assemblages to fire is warranted.

The DPIPWE guidelines also specify maximum intervals but there is low confidence in these values because very little is known about the long-term consequences of not burning for soils, plants and animals. There is probably no practical value in setting a maximum fire interval for invertebrates because there is currently no evidence of any taxon requiring early successional stages and because unplanned fires are likely to occur in the absence of planned burns (King et al. 2006).

Season of Burn

There has been very little study on the effect of season of burn on moorland ecosystems (Balmer and Storey 2010; Driessen 2010; Storey 2010). No season of burn was identified as optimal in the DPIPWE guidelines other than excluding summer fire because of the risks of escapes into fire sensitive vegetation and loss of peat. Defining an optimal season of burn for invertebrates can be difficult because of variation in species requirements (New 2014). Season of burn was not specifically addressed in the present study. A BACI study was commenced as part of the present

study comparing the response of invertebrates to spring and autumn burns but this experiment was severely compromised by an escaped planned burn (unpublished data). In Chapter 3, the season of burn for the last fire was known for many sites, with most burnt in spring or autumn and some during summer, and nearly all sites had been subjected to summer wildfire at least once since the 1930s. Many of the old regrowth (25–57 years) lowland sites burnt by summer wildfires had invertebrate assemblage compositions that were more similar to old regrowth sites burnt by low intensity planned burns conducted in spring or autumn than young regrowth sites (Chapter 3, Fig. 3.2). I expect that season of burn will have a limited effect on the return moorland invertebrates to the pre-fire state given that moorland vegetation is a fire-prone community and follows a highly deterministic succession after low-intensity fire (Chapters 3 and 5; Balmer and Storey 2010). However, this hypothesis requires testing and New (2014) warns against burning at the same time of year every year as this may lead to selection of certain species and communities over others.

Intensity

Fire intensity relates to the amount of heat released per unit time and is influenced by the amount of fuel and rate of spread. It has a strong influence on the impact of, and ability to manage fires. The DPIPWE guidelines recommend low intensity planned burns (<500 kWm^{-1}) primarily to minimise soil loss and the escape of fires beyond their intended boundaries. Ideally, the intensity should be low enough to leave thatch on the ground after the burn to protect post-fire soils and enhance post-fire germination rates (Balmer and Storey 2010; Storey 2010). The presence of thatch may also be beneficial to ground-active invertebrates by either providing post-fire shelter or perhaps because it implies that the intensity was low enough for more invertebrates to survive the direct effects of the fire. The effect of fire intensity on moorland fauna requires investigation.

Spatial extent and distribution of planned burns

The DPIPWE guidelines provide general comments in relation to the spatial context of planned burns. In particular they recommend that there should be a mosaic of

burnt and unburnt areas at a local and regional scale. Spatial variability in fire regimes is considered important for the maintenance of biodiversity. Recommendations for fire mosaics are often the conclusion of studies that have found that different species prefer different stages of the post fire succession (e.g. Fox and McKay 1981; Moretti et al. 2004; Brennan et al. 2006). However the influence of the spatial arrangements of fire regimes on the persistence of species is poorly understood (Clarke 2008; Driscoll et al. 2010) and this is acknowledged in DPIPWE's guidelines. A recent study suggests that maximising pyrodiversity by having an equal mix of age classes did not necessarily promote vertebrate biodiversity, with older vegetation disproportionally important for birds, reptiles and mammals (Kelly et al. 2015). While further investigation is warranted, such studies will prove difficult partly because of the scale of the studies and partly because fire-related variability is easily confounded with other environmental variability at these scales (Keith et al. 2002b). Keith et al. (2002b) also caution that many practical applications of mosaic burning result in rigid rotational block burning programs that are ill-conceived for their intended purpose and reflect logistics of controlling fire more than ecological criteria relevant to the target organisms. To investigate characteristics of mosaic burning such as patch size, connectivity, and composition of time-since-fire classes, landscape-scale space-for-time substitution studies have been recommended (Clarke 2008).

Invertebrate monitoring

Adaptive fire management requires monitoring and evaluation, with future actions modified in light of new evidence (Clarke 2008). Monitoring invertebrates can be an expensive and time consuming process. In the present study, two approaches were investigated that can reduce the time and expense associated with invertebrate monitoring. Higher taxa were found to be effective surrogates of species-level identification; and environmental variables, such as vegetation density, were found to be useful predictors of invertebrate assemblage compositions change, at least for ground- and foliage-active invertebrates. The successful use of these surrogates will depend on understanding their limitations, the objectives of the monitoring and the value of results versus resources saved.

7.6 Further Research

Despite important progress over recent decades, fire management practices continue to be based on limited biodiversity conservation information, particularly for fauna. Addressing the knowledge gaps requires a combination of targeted research and adaptive management together with financial investment and research commitment (Clarke 2008; Driscoll et al. 2010). In a recent review, Driscoll et al. (2010) identified three broad knowledge gaps that currently limit our ability to make informed decisions about fire management for biodiversity conservation and that are relevant to improving fire management in Tasmanian moorlands:

1. Developing a mechanistic understanding of species responses to fire regimes
2. Knowledge of how the spatial and temporal arrangement of fire regimes influences biota, and
3. Understanding interactions of fire regimes with other processes.

These conclusions echo those of earlier reviews (Whelan 1995; Keith et al. 2002a; Whelan et al. 2002; Clarke 2008). Developing mechanistic models requires autecological data which is frequently lacking for many invertebrate species and which limited the assessment of species traits in the present study. Assessing the response of plants and animals to differing fire regimes at a variety of spatial and temporal scales will be challenging because of the scale of the studies and because fire-related variability is easily confounded with other environmental variability at these scales (Keith et al. 2002b). Investigation of the third knowledge gap will depend on processes that are operating in an ecosystem. Among the three 'stand out' processes identified by Driscoll et al. (2010), only climate change is relevant to fire management in buttongrass moorlands, with predictions of increased fire incidence and severity under projected climate change scenarios for western Tasmania (DPIPWE 2010). Importantly, Driscoll et al. (2010) identify a research agenda to maximise the rate of learning in this difficult field. This includes measuring responses at a species level, building capacity to implement natural

experiments, undertaking simulation modelling, and judicious application of experimental approaches.

Monitoring of biodiversity within the context of adaptive management is complex and not well-developed (Keith et al 2002; Clarke 2008). It is not currently undertaken in buttongrass moorlands. Most monitoring projects fail because of insufficient resourcing, flawed design and lack of commitment (Keith and Tozer 1997). Investigating knowledge gaps and monitoring for adaptive management in buttongrass moorland presents challenges and opportunities because much of the community occurs in remote locations, requiring overnight walking or helicopter access, the invertebrate fauna is poorly documented and autecological data are available for a very small proportion of taxa. Buttongrass moorland has potential as a model community to investigate important fire research questions because of (i) limited structural and floristic diversity, (ii) the ease of conducting planned burns in this vegetation type compared to forest, and (iii) its large areas in reserves.

Further fire management research on buttongrass moorland fauna should focus on soil-active fauna because previous work (Green 2009) suggests that they may be more sensitive to fire than ground- and foliage-active fauna. Research should also target 'Sparse Buttongrass Moorlands on Slopes' because it is the most abundant moorland class in southwest Tasmania (Corbett and Balmer 2007) and because fire can result in substantial soil losses in this moorland (di Folco and Kirkpatrick 2011; DPIPWE 2015). Moorlands also support a diversity of freshwater fauna in adjacent streams and rivers, as well as in pools and burrows within the moorland proper. In particular, moorlands support a highly distinct assemblage of freshwater crayfish (*Ombrastacoides* spp. and *Spinasticoides* spp.) whose burrows provide habitat for a fauna known collectively as pholeteros (Lake 1977). The impact of fire on water-active fauna in moorlands has not been studied.

To conserve the full range of biodiversity, Keith et al. (2002b) recognised the need for fire management plans that address critical elements such as populations of threatened species. Three fauna species listed as threatened under the Tasmanian *Threatened Species Protection Act 1995* are restricted to buttongrass moorlands; two species of Trichoptera, *Taskiria mccubbini* and *Taskiropsyche laucustris* and a primitive syncarid crustacean, *Allanaspides hickmani* (Driessen

2008). Research is needed on fire regime requirements for these species. This will be challenging for the caddisflies because they have been so rarely recorded (Jackson 2000).

An overall better understanding of moorland invertebrate fauna is required to identify the extent to which species are restricted to this vegetation type. Only 16% of the species recognised in Chapters 2 and 3 could be assigned a scientific name (Appendix 1) and virtually all of these are widely occurring taxa that were readily identified by specialists. Although eleven new taxa have been described (Appendix 1), proper description of the moorland fauna was beyond the scope of the present study and beyond the time, resources and expertise of the specialists involved. Assessing the extent to which species are restricted to moorland will also require surveys of moorlands considered to be climax communities and vegetation types occurring adjacent to moorlands, particularly scrub. This information will not only assist with understanding the responses of moorland fauna to fire (i.e. does adjacent vegetation provide a refuge for recolonisation) but also contribute to further understanding the significance of this world heritage ecosystem.

References

Abbott, I. 1984. Changes in the abundance and activity of certain soil and litter fauna in the jarrah forest of Western Australia after a moderate intensity fire. *Soil Research* 22: 463-469.

Abbott, I., van Heurch, P., and Wong, L. 1984. Responses to long-term fire exclusion: physical, chemical and faunal features of litter and soil in a Western Australian forest. *Australian Forestry* 47: 237–242.

Abraham, B. J. 1983. Spatial and Temporal Patterns in a Sagebrush Steppe Spider Community (Arachnida, Araneae). *Journal of Arachnology* 11: 31-50.

Andersen, A. N. 1995. A classification of Australian ant communities, based on functional groups which parallel plant life-forms in relation to stress and disturnabce. *Journal of Biogeography* 22: 15–29.

Andersen, A. N., and Müller, W. J. 2000. Arthropod responses to experimental fire regimes in an Australian tropical savanna: ordinal-level analysis. *Austral Ecology* 25: 199–209.

Andersen, A. N., and Yen, A. L. 1985. Immediate effects of fire on ants in the the semi-arid malle region of north-western Victoria. *Australian Journal of Ecology* 10: 25–30.

Anderson, D. R., and Burnhan, K. P. 2002. Avoiding pitfalls when using information-theoretic methods. *Journal of Wildlife Management* 66: 912–918.

Anderson, M. J. 2001. A new method for non-parametric multivariate analysis of variance. *Austral Ecology* 26: 32–46.

Anderson, M. J., Gorley, R. N., and Clarke, K. R. 2008. *PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods*. PRIMER-E Ltd, Plymouth.

Anderson, M. J., and Robinson, J. 2003. Generalised discriminant analysis based on distances. *Australian and New Zealand Journal of Statistics* 45: 301–318.

Anu, A., Sabu, T. K., and Vineesh, P. J. 2009. Seasonality of litter insects and relationship with rainfall in a wet evergreen forest in south Western Ghats. *Journal of Insect Science* 9: 1–10.

Arkel, B. 1995. Small mammal secondary succession in buttongrass moorlands. MSc Thesis. University of Tasmania, Hobart, Tasmania.

Aubin, I., Venier, L., Pearce, J., and Moretti, M. 2013. Can a trait-based multi-taxa approach improve our assessemnt of forest management impact on biodiversity. *Biodiversity and Conservation* 22: 2957–2975.

Badejo, M. A. 1990. Seasonal abundance of soil mites (Acarina) in two contrasting environments. *Biotropica* 22: 382–390.

Balmer, J., and Storey, D. 2010. A review of vegetation responses to fire in buttongrass moorland. Pages 22–29 *in* J. Balmer, editor. *Proceedings of the 2007 Buttongrass Moorland Management Workshop*. Department of Primary Industries, Parks, Water and Environment, Hobart, Tasmania.

Balmer, J., Whinam, J., Kelman, J. B., Kirkpatrick, J. B., and Lazarus, E. 2004. Floristic Values of the Tasmanian Wilderness World Heritage Area. 2004/3, Department of Primary Industries, Water and Environment, Hobart. 129 pages.

Balmford, A., Jayasuriya, A. H. M., and Green, M. J. B. 1996. Using higher-taxon richness as a surrogate for species richness: II. Local applications. *Proceedings of the Royal Society of London B* 263: 1571-1575.

Baribeau, L., and Maire, A. 1983. Abundance and seasonal distribution of Tabanidae in a temperate and in a subartic locality of Quebec. *Mosquito News* 43: 135–142.

Barratt, B. I. P., Ferguson, C. M., Barton, D. M., and Johnstone, P., D,. 2009. Impact of fire on tussock grasland invertebrate populations. Science for Conservation 291. Department of Conservation, Wellington. 75 pages.

Barrow, L., and Parr, C. L. 2008. A preliminary investigation of temporal patterns in semiarid ant communities: variation with habitat type. *Austral Ecology* 33: 653-662.

Basset, Y., Mavoungou, J. F., Mikissa, J. B., Missa, O., Miller, S. E., Kitching, R. L., and Alonso, A. 2004. Discriminatory power of different arthropod data sets for the biological monitoring of anthropogenic disturbance in tropical forests. *Biodiversity and Conservation* 13: 709-732.

Bell, H. L. 1985. Seasonal variation and the effects of drought on the abundance of arthropods in savanna woodland on the Northern Tablelands of New South Wales. *Australian Journal of Ecology* 10: 207–221.

Bess, E. C., Parmenter, R. R., McCoy, S., and Molles Jr, M. C. 2002. Responses of a riparian forest-floor arthropod community to wildfire in the middle Rio Grande Valley, New Mexico. *Environmental Entomology* 31: 774–784.

Bevilacqua, S., Claudet, J., and Terlizzi, A. 2013. Best practicable aggregation of species: a step forward for species surrogacy in environmental assessment and monitoring. *Ecology and Evolution* 3: 3780–3793.

Bevilacqua, S., Terlizzi, A., Claudet, J., Fraschetti, S., and Boero, F. 2012. Taxonomic relatedness does not matter for species surrogacy in the assessment of community responses to environmental drivers. *Journal of Applied Ecology* 49: 357–366.

Biaggini, M., Consorti, R., Dapporto, L., Dellacasa, M., Paggetti, E., and Corti, C. 2007. The taxonomic level order as a possible tool for rapid assessment of Arthropod diversity in agricultural landscapes. *Agriculture Ecosystems & Environment* 122: 183-191.

Bock, C. E., and Bock, J. H. 1991. Response of grasshoppers (Orthoptera: Acrididae) to wildlife in a southeastern Arizona grassland. *American Midland Naturalist* 125: 162–167.

Bowman, D. M. J. S. 2008. Is global environmental change the end game for prehistoric legacies? The parallel cases of fire-maintained vegetation mosaics in southwest Tasmania and central Australia. *Australian Plant Conservation* 16: 6–8.

Bowman, D. M. J. S., Maclean, A. R., and Crowden, R. K. 1986. Vegetation-soil relations in the lowlands of southwest Tasmania. *Australian Journal of Ecology* 11: 141-153.

Bradstock, R. A., Keith, D. A., and Auld, T. D. 1995. Fire and conservation: Imperatives and constraints on managing for diversity. Pages 323-333 *in* R. A. Bradstock, T. D. Auld, D. A. Keith, R. T. Kingsford, D. Lunney, and D. P. Sivertsen, editors. *Conserving Biodiversity: Threats and Solutions*. Surrey Beatty and Sons in association with NSW National Parks and Wildlife Service, New South Wales.

Brennan, K. E. C., Ashby, L., Majer, J. D., Moir, M. L., and Koch, J. M. 2006. Simplifying assessment of forest management practices for invertebrates:

How effective are higher taxon and habitat surrogates for spiders following prescribed burning? *Forest Ecology and Management* 231: 138–154.

Bridle, K. P., Cullen, P., and Russell, M. 2003. Peatland hydrology, fire management and Holocene fire regimes in southwest Tasmanian blanket bogs. Department of Primary Industries and Water, Hobart, Tasmania. 92 pages.

Brown, M. J., Brown, P. B., Bryant, S. J., Horwitz, P., McQuillan, P. B., Nielsen, E., Rounsevell, D. E., Smith, S. J., and Richardson, A. M. M. 1993. Buttongrass moorland ecosystems. Pages 101–108 *in* S. J. Smith and M. R. Banks, editors. *Tasmanian Wilderness—World Heritage Values*. Royal Society of Tasmania, Hobart, Tasmania.

Bryant, S. 1991. The ground parrot, Pezoporus wallicus, in Tasmania: distribution, density and conservation status. Wildlife Division Scientific Report 91/1, Department of Parks, Wildlife and Heritage, Hobart, Tasmania.

Bullock, S. H., Mooney, H. A., and Medina, E. 1998. *Seasonally Dry Tropical Forests*. Cambridge University Press, Melbourne.

Burnham, K. P., and Anderson, D. R. 2002. *Model selection and multi-model inference: a practical information theoretic approach, 2nd edn*. Springer, New York.

Burnham, K. P., Anderson, D. R., and Huyvaert, K. P. 2011. AIC model selection and multimodel inference in behavioral ecology: some background, observations and comparisons. *Behavioural Ecology and Sociobiology* 65: 23–35.

Cardoso, P., Silva, I., De Oliveira, N. G., and Serrano, A. R. M. 2007. Seasonality of spiders (Araneae) in Mediterranean ecosystems and its implications in the optimum sampling period. *Ecological Entomology* 32: 516–526.

Chatzaki, M., Lymberakis, P., Mitov, P., and Mylonas, M. 2009. Phenology of Opiliones on an altitudinal gradient on Lefka Ori Mountains, Crete, Greece. *Journal of Arachnology* 37: 139–146.

Chaudhry, T. A. 2010. Avifaunal ecology and responses to post-fire succession of buttongrass moorlands in the Tasmanian Wilderness World Heritage Area. PhD Thesis. University of Tasmania, Hobart.

Clarke, K. R., and Ainsworth, M. 1993. A method for linking multivariate community structure to environmental variables. *Marine Ecology Progress Series* 92: 205–219.

Clarke, K. R., and Gorley, R. N. 2006. *PRIMER v6: User Manual/Tutorial*. PRIMER-E Ltd, Plymouth.

Clarke, K. R., and Warwick, R. M. 2001. *Change in marine communities: an approach to statistical analysis and interpretation, 2nd edn*. PRIMER-E, Plymouth.

Clarke, M. F. 2008. Catering for the needs of fauna in fire management: science or just wishful thinking? *Wildlife Research* 35: 385–394.

Coleman, T. W., and Rieske, L. K. 2006. Arthropod response to precription burning at the soil-litter interface in oak-pine forests. *Forest Ecology and Management* 233: 52–60.

Cook, W. M., and Holt, R. D. 2006. Fire frequency and mosaic burning effects on a tallgrass prairie ground beetle assemblage. *Biodiversity and Conservation* 15: 2301-2323.

Corbett, S., and Balmer, J. 2007. Buttongrass moorland in Tasmania — what and where? *Australasian Plant Conservation* 16: 3–5.

Cruz-Sánchez, M. A., Asís, J. D., Gayubo, S. F., Tormos, J., and Gonzáles, J. A. 2011. The effects of wildfire on Spheciformes wasp community structure: the importance of local habitat conditions. *Journal of Insect Conservation* 15: 487–503.

Danks, H., V. 2006. Key themes in the study of seasonal adaptations in insects II. Life-cycle patterns. *Applied Entomology and Zoology* 41: 1–13.

Danks, H., V. 2007. The elements of seasonal adaptations in insects. *Canadian Entomologist* 139: 1–44.

Denlinger, D. L. 1980. Seasonal and annual variation of insect abundance in the Nairobi National Park, Kenya. *Biotropica* 12: 100–106.

di Folco, M.-B. 2007. Tasmanian Organic Soils. PhD Thesis. University of Tasmania, Hobart.

di Folco, M.-B., and Kirkpatrick, J. B. 2011. Topographic variation in burning-induced loss of carbon from organic soils in Tasmanian moorlands. *Catena* 87: 216-225.

di Folco, M.-B., and Kirkpatrick, J. B. 2013. Organic soils provide evidence of spatial variation in human-induced vegetation change following European occupation of Tasmania. *Journal of Biogeography* 40: 197–205.

Didham, R. K., Hammond, P. M., Lawton, J. H., Eggleton, P., and Stork, N. E. 1998. Beetle species responses to tropical forest fragmentation. *Ecological Monographs* 68: 295-323.

DPIPWE. 2010. Vulnerability of Tasmania's Natural Environment to Climate Change: An Overview. Department of Primary Industries, Parks, Water and Environment, Hobart. 79 pages.

DPIPWE. 2011. Southern Region Strategic Fire Management Plan. Department of Primary Industries, Parks, Water and Environment, Hobart, Tasmania. 88 pages.

DPIPWE. 2012. Northwest Region Strategic Fire Management Plan. Department of Primary Industries, Parks, Water and Environment, Hobart, Tasmania. 85 pages.

DPIPWE. 2015. Fire Regimes for Nature Conservation in the Tasmanian Wilderness World Heritage Area. . Natural and Cultural Heritage Division, Department of Primary Industries Parks Water and Environment, Hobart, Nature Conservation Report Series 15/2.

Driessen, M. 2010. A review of fauna responses to fire in buttongrass moorland. Pages 30–35 *in* J. Balmer, editor. *Proceedings of the 2007 Buttongrass Moorland Management Workshop*. Department of Primary Industries, Parks Water and Environment, Hobart, Tasmania.

Driessen, M. M. 1999. Effects of fire on the broad-toothed mouse, *Mastacomys fuscus*, and other small mammals in buttongrass moorlands of western Tasmania - preliminary findings. Pages 119–126 *The Proceedings of the Australian Bushfire Conference, Albury Australia 7-9 July 1999*.

Driessen, M. M. 2006. The Fauna of Buttongrass Moorland. *The Tasmanian Naturalist* 128: 37–51.

Driessen, M. M. 2008. Buttongrass moorland fauna. *Australian Plant Conservation* 16: 20–22.

Driessen, M. M., and Greenslade, P. 2004. Effect of season, location and fire on Collembola communities in buttongrass moorlands, Tasmania. *Pedobiologia* 48: 631–642.

Driessen, M. M., Kirkpatrick, J. B., and McQuillan, P. B. 2013. Shifts in Composition of Monthly Invertebrate Assemblages in Moorland Differed Between Lowland and Montane Locations but not Fire-Ages. *Environmental Entomology* 42: 58-73.

Driscoll, D. A., and Henderson, M. K. 2008. How many common reptile species are fire specialists? A replicated natural experiment highlights the predictive weakness of a fire succession model. *Biological Conservation* 141: 460-471.

Driscoll, D. A., Lindenmayer, D. B., Bennett, A. F., Bode, M., Bradstock, R. A., Cary, G. J., Clarke, M. F., Dexter, N., Fensham, R., Friend, G., Gill, M., James, S., Kay, G., Keith, D. A., MacGregor, C., Russell-Smith, J., Salt, D., Watson, J. E. M., Williams, R. J., and York, A. 2010. Fire management for biodiversity conservation: Key research questions and our capacity to answer them. *Biological Conservation* 143: 1928-1939.

Driscoll, D. A., Smith, A. L., Blight, S., and Maindonald, J. 2012. Reptile responses to fire and the risk of post-disturbance sampling bias. *Biodiversity and Conservation* 21: 1607-1625.

Driscoll, D. A., and Weir, T. 2005. Beetle responses to habitat fragmentation depend on ecological traits, habitat condition, and remnant size. *Conservation Biology* 19: 182–194.

Dyer, R., Cobiac, M., Cafe, L., and Stockwell, T. 1997. Developing Sustainable Pasture Management Practices for the Semi-Arid Tropics of the Northern Territory. Final report for the Developing Sustainable Pasture and Management practices (NTA022) Project. Northern Territory Department of Primary Industries and Fisheries, Katherine.

Ellis, D. 1985. Taxonomic sufficiency in pollution assessment. *Marine Pollution Bulletin* 16: 442–461.

Evans, E. W. 1984. Fire as a natural disturbance to grasshopper assemblages of tallgrass prairie. *Oikos* 43: 9–16.

Ewers, R. M., and Didham, R. K. 2006. Confounding factors in the detection of species responses to habitat fragmentation. *Biological Reviews* 81: 117-142.

Ferguson, S. H. 2004. Influence of edge on predator–prey distribution and abundance. *Acta Oecologia* 25: 111-117.

Fox, B. J. 1982. Fire and mammalian secondary succession in an Australian coastal heath. *Ecology* 63: 1332–1341.

Fox, B. J., and McKay, G. M. 1981. Small mammal responses to pyric successional changes in eucalypt forest. *Australian Journal of Ecology* 6: 29–41.

Friend, G. R. 1993. Impact of fire on small vertebrates in mallee woodlands and heathlands of temperate Australia: A review. *Biological Conservation* 65: 99-114.

Friend, G. R. 1995a. Fire and invertebrate - a review of research methodology and the predictability of post-fire response patterns. *CALMScience Supplement* 4: 165–174.

Friend, G. R. 1995b. Fire and invertebrates: a review of research methodology and the predictability of post-fire response patterns. *CALMScience Supplement* 4: 165–174.

Friend, G. R., and Williams, M. R. 1996. Impact of fire on invertebrate communities in mallee-heath shrublands of southwestern Australia. *Pacific conservation biology* 2: 244–267.

Friend, J. A., and Richardson, A. M. M. 1986. The Biology of Terrestrial Amphipods. *Annual Review of Entomology* 31: 25–48.

Gaston, K. J., and Williams, P. H. 1993. Mapping the worlds species - the higher taxon approach. *Biodiversity Letters* 1: 2–8.

Gill, A. M., Bradstock, R. A., and Williams, J. E. 2002. Fire regimes and biodiversity: legacy and vision. Pages 430–446 *in* R. A. Bradstock, J. E. Williams, and A. M. Gill, editors. *Flammable Australia*. Cambridge University Press, Cambridge.

Giller, P. S. 1996. The diversity of soil communities, the 'poor man's tropical rainforest'. *Biodiversity and Conservation* 5: 135–168.

Gosper, C. R., Pettit, M. J., Andersen, A. N., Yates, C. J., and Prober, S. M. 2015. Multi-century dynamics of ant communities following fire in Mediterranean-

climate woodlands: Are changes congruent with vegetation succession? *Forest Ecology and Management* 342: 30-38.

Green, D. 2008. The diversity of soil mites in Tasmanian buttongrass moorland in relation to vegetation age. *Australasian Plant Conservation* 16: 24–26.

Green, D. 2009. The soil mites of buttongrass moorland (Tasmania) and their response to fire as a management tool. Pages 179–183 *in* M. W. Sabelis and J. Bruin, editors. *Trends in Acarology*. Springer, Amsterdam.

Greenslade, P., and Driessen, M. M. 1999. The effect of fire on epigaeic arthropods in buttongrass moorland in Tasmania. Pages 82–89 *in* W. Ponder and D. Lunney, editors. *The Other 99%. The Conservation and Biodiversity of Invertebrates*. Transactions of the Royal Zoological Society of New South Wales.

Greenslade, P., and Smith, D. 1999. The epigaic arthropod fauna of buttongrass moorland in the Tasmanian Wilderness World heritage Area. Pages 90–94 *in* W. Ponder and D. Lunney, editors. *The Other 99%. The Conservation and Biodiversity of Invertebrates*. Transactions of the Royal Zoological Society of New South Wales.

Greenslade, P., and Smith, D. 2010. Short term effects of wild fire in invertebrates in coastal heathland in southeastern Australia. *Pacific conservation biology* 16: 1–10.

Grimbacher, P. S., and Stork, N. E. 2009. Seasonality of a diverse beetle assemblage inhabiting lowland tropical rain forest in Australia. *Biotropica* 41: 328-337.

Gullan, P. J., and Cranston, P. S. 2005. *The insects: an outline of entomolgy*. Blackwell Publishing, Melbourne, Australia.

Gullan, P. J., and Cranston, P. S. 2010. *The Insects: An Outline of Entomology, 4th Edition*. Wiley-Blackwell, London.

Gutiérrez, D., and Menénendez. 1998. Phenology of butterflies along an altitudinal gradient in northern Spain. *Journal of Zoology* 244: 249–264.

Haddad, N. M., Tilman, D., Haarstad, J., Ritchie, M. E., and Knops, J. M. H. 2001. Contrasting effects of plant richness and composition on insect communities: a field experiment. *American Naturalist* 158: 17–35.

Hale, W. G. 1966. A population study of moorland Collembola. *Pedobiologia* 6: 65–99.

Hallet, C. S., Valesini, F. J., and Clarke, K. R. 2012. A method for selecting health index metrics in the absence of independent measures of ecological condition. *Ecological Indicators* 19: 240–252.

Hanula, J. L., and Wade, D. D. 2003. Influence of long-term dormant-season burning and fire exclusion on ground-dwelling arthropod populations in longleaf pine flatwoods ecosystems. *Forest Ecology and Management* 175: 163-184.

Heino, J., and Soininen, J. 2007. Are higher taxa adequate surrogates for species-level assemblage patterns and species richness in stream organisms? *Biological Conservation* 137: 78-89.

Hochkirch, A., and Adorf, F. 2007. Effects of prescribed burining and wildfires on orthoptera in Central European peat bogs. *Environmental Conservation* 34: 225–235.

Hoffmann, B. D. 2003. Responses of ant communities to experimental fire regimes on rengelands in the Victoria River District of the Northern Territory. *Austral Ecology* 28.

Holz, A., Wood, S. W., Veblen, T. T., and Bowman, D. 2015. Effects of high-severity fire drove the population collapse of the subalpine Tasmanian endemic conifer Athrotaxis cupressoides. *Global Change Biology* 21: 445–458.

Hopkin, S. P. 1997. *Biology of the Springtails*. Oxford University Press, Oxford.

Hulbert, L. C. 1988. Causes of fire effects in tallgrass prairie. *Ecology* 69: 46-58.

Hutchinson, G. E. 1959. Homage to Santa Rosalia or why are there so many kinds of animals *American Naturalist* 93: 145–159.

Jackson, J. 2000. Threatened Trichoptera (caddisflies) from the Tasmanian Wilderness World Heritage Area. *Papers and proceedings of the Royal Society of Tasmania* 134: 55–62.

Jackson, W. D. 1968. Fire, air, water and earth – an elemental ecology of Tasmania. *Proceedings of the Ecological Society of Australia* 3: 9–16.

Jackson, W. D. 1999. The Tasmanian Environment. Pages 11–38 *in* B. R. Reid, R. S. Hill, M. J. Brown, and M. J. Hovenden, editors. *Vegetation of Tasmania*. Australian Biological Resources Study, Environment Australia, Canberra.

Jarman, S. J., Kantvilas, G., and Brown, M. J. 1988. Buttongrass Moorland in Tasmania. Research Report 2, Tasmanian Forest Research Council Inc., Hobart, Tasmania. 155 pages.

Joern, A., and Laws, A. N. 2013. Ecological Mechanisms Underlying Arthropod Species Diversity in Grasslands. *Annual Review of Entomology* 58: 19–36.

Johnson, E. A., and Miyanishi, K. 2008. Testing the assumptions of chronosequences in succession. *Ecology Letters* 11: 419–431.

Johnson, K. J., and Marsden-Smedley, J. B. 2001. Fire history of the northern part of the Tasmanian Wilderness World Hertigae Area and its associated regions. *Papers and proceedings of the Royal Society of Tasmania* 136: 145–152.

Keith, D. A. 1996. Fire-driven extinction of plant populations: a synthesis of theory and review of evidence from Australian vegetation. *Proceedings of the Linnean Society of New South Wales* 116: 37–78.

Keith, D. A., and Bradstock, R. A. 1994. Fire and competition in Australian heath: a conceptual model and field investigation. *Journal of Vegetation Science* 5.

Keith, D. A., McCaw, W. L., and Whelan, R. J. 2002a. Fire regimes in Australian heathlands and their effects on plants and animals. Pages 199-237 *in* R. A. Bradstock, J. E. Williams, and A. M. Gill, editors. *Flammable Australia*. Cambridge University Press, Cambridge.

Keith, D. A., Williams, J. E., and Woinarski, C. Z. 2002b. Fire management and biodiversity conservation: key approaches and principles. Pages 401–425 *in* R. A. Bradstock, J. E. Williams, and A. M. Gill, editors. *Flammable Australia*. Cambridge University press, Cambridge.

Kelly, L. T., Bennett, A. F., Clarke, M. F., and McCarthy, M. A. 2015. Optimal fire histories for biodiversity conservation. *Conservation Biology* 29: 473-481.

Kim, J. W., and Jung, C. 2013. Ecological disturbance of soil oribatid mite communities after fire disturbance. *Journal of Ecology and Environment* 36: 117–123.

Kim, K. C. 1993. Biodiversity, conservation and inventory - why insects matter. *Biodiversity and Conservation* 2: 191-214.

King, K. J., Cary, G. J., Bradstock, R. A., Chapman, J., Pyrke, A., and Marsden-Smedley, J. B. 2006. Simulation of prescribed burning strategies in south-west

Tasmania, Australia: effects on unplanned fires, fire regimes, and ecological management values. *International Journal of Wildland Fire* 15: 527-540.

Kishimoto-Yamada, K., Itioka, T., Sakai, S., and Ichie, T. 2010. Seasonality in light-attracted chrysomelid populations in a Bornean rainforest. *Insect Conservation and Diversity* 3: 266-277.

Knops, J. M. H., Tilman, D., Haddad, N. M., Naeem, S., Mitchell, C. E., Haarstad, J., Ritchie, M. E., Howe, K. M., Reich, P. B., Siemann, E., and Groth, J. 1999. Effects of plant species richness on invasion dynamics, disease outbreaks, insect abundances and diversity. *Ecology Letters* 2: 286–293.

Koricheva, J., Mulder, C. P. H., Schmid, B., Joshi, J., and Huss-Danell, K. 2000. Numerical responses of different trophic groups of invertebrates to manipulations of plant diversity in grasslands,. *Oecologia* 125: 275–282.

Kruess, A., and Tscharntke, T. 1994. Habitat fragmentation, species loss, and biological control. *Science* 264: 1581-1584.

Lake, P. S. 1977. Pholeteros: the fauna assemblage found in crayfish burrows. *Australian Society for Limnology Newsletter* 15: 57–60.

Langlands, P. R., Brennan, K. E. C., and Ward, B. 2012. Is the reassembly of an arid spider assemblage following fire deterministic? *Austral Ecology* 37: 429-439.

Lawton, J. H. 1983. Plant architecture and the diversity of phytophagus insects. *Annual Review of Entomology* 28: 23–39.

Lek-Ang, S., and Deharveng, L. 2002. Seasonal variations of Collembola (Hexapoda) assemblages in damp riparian habitats. *Archiv für Hydrobiologie* 153: 635–655.

Lenat, D. R., and Resh, V. H. 2001. Taxonomy and stream ecology: the benfits of genus- and species-level identifications. *Journal of the North American Benthological Society* 20: 287–298.

Letnic, M., Dickman, C. R., Tischler, M. K., Tamayo, B., and Beh, C. L. 2004. The responses of small mammals and lizards to post-fire succession and rainfall in arid Australia. *Journal of Arid Environments* 59: 85-114.

Lovell, S., Hamer, M., Slotow, R., and Herbert, D. 2007. Assessment of congruency across invertebrate taxa and taxonomic levels to identify potential surrogates. *Biological Conservation* 139: 113-125.

Lowman, M. D. 1982. Seasonal variation in insect abundance among three Australian rain forests, with particular reference to phytophagus types. *Australian Journal of Ecology* 7: 353–361.

Majer, J. D. 1984. Short-term responses of soil and litter invertebrates to a cool autumn burn in Jarrah (Eucalyptus marginata) forest in Western Australia. *Pedobiologia* 26: 229–247.

Mandelik, Y., Dayan, T., Chikatunov, V., and Kravchenko, V. 2007. Reliability of a higher-taxon approach to richness, rarity, and composition assessments at the local scale. *Conservation Biology* 21: 1506-1515.

Maraun, M., and Scheu, S. 2000. The structure of oribatid mite communities (Acari, Oribatida): patterns, mechanisms and implications for future research. *Ecography* 23: 374-383.

Marra, J. L., and Edmonds, R. L. 1998. Effects of coarse woody debris and soil depth on the density and diversity of soil invertebrates on clearcut and forested sites on the Olympic Peninsula, Washington. *Environmental Entomology* 27: 1111-1124.

Marsden-Smedley, J. B. 1998. Changes in southwestern Tasmanian fire regimes since the early 1800s. *Papers and proceedings of the Royal Society of Tasmania* 132: 15–29.

Marsden-Smedley, J. B. 2009. Planned burning in Tasmania: operational guidelines and review of current knowledge. Parks and Wildlife Service, Hobart, Tasmania. 93 pages.

Marsden-Smedley, J. B., and Kirkpatrick, J. B. 2000. Fire Management in Tasmania's Wilderness World Heritage Area: Ecosystem restoration using Indigenous-style fire regimes. *Ecological Management and Restoration* 1: 195–203.

Masaki, S. 1983. Introduction. Pages 1–6 *in* V. K. Brown and I. Hodek, editors. *Diapause and Life Cycle Strategies in Insects*. Dr W. Junk Publishers, The Hague.

Masaki, S., and Wipking, W. 1994. Insect life-cycle polymorphism: Introduction. Pages 1–4 *in* H. Danks, V, editor. *Insect Life-cycle Polymorphism: Theory, Evolution and Ecological Consequences for Seasonality and Diapause Control*. Kluwer Academic Publishers, Dordrectht.

Masteller, E. C. 1993. Comparison of tropical and temperate emergence phenology of aquatic insects from Puerto Rico and pennsylvania. *Journal of the Kansas Entomological Society* 66: 192–199.

Masters, P. 1993. The effects of fire-driven succession and rainfall on small mammals in spinifex grassland at Uluru National Park, Northern Territory. *Wildlife Research* 20: 803-813.

Mateos, E., Santos, X., and Pujade-Villar, J. 2011. Taxonomic and Functional Responses to Fire and Post-Fire Management of a Mediterranean Hymenoptera Community. *Environmental Management* 48: 1000-1012.

Matsuda, T., Turschak, G., Brehme, C., Rochester, C., Mitrovich, M., and Fisher, R. 2011. Effects of Large-Scale Wildfires on Ground Foraging Ants (Hymenoptera: Formicidae) in Southern California. *Environmental Entomology* 40: 204-216.

McArdle, B. H., and Anderson, M. J. 2001. Fitting multivariate models to community data: a comment on distance -based redundancy analysis. *Ecology* 82: 290–297.

McCraken, D. I. 1994. A fuzzy classfication of moorland ground beetle (Cleoptera: Carabidae) and plant communities. *Pedobiologia* 38: 12–27.

McFarland, D. 1988. The composition, microhabitat use and response to fire of the avifauna of sub-tropical heatlands in Cooloola National Park, Queensland. *Emu* 88: 249-257.

McGavin, G. C. 2000. *Insects, spiders and other terrestrial arthropods*. Dorling Kindersley Limited, London.

Melbourne, B. A. 1999. Bias in the effect of habitat structure on pitfall traps: An experimental evaluation. *Australian Journal of Ecology* 24: 228-239.

Mitchell, M. J. 1978. Vertical and horizontal distributions of oribatid mites (Acari: Cryptostigmata) in an aspen woodland soil. *Ecology* 59: 516–525.

Monamy, V., and Fox, B. J. 2000. Small mammal succession is determined by vegetation density rather than time elapsed since disturbance. *Austral Ecology* 25: 580–587.

Monamy, V., and Fox, B. J. 2010. Responses of two species of heathland rodents to habitat manipulation: Vegetation density thresholds and the habitat accommodation model. *Austral Ecology* 35: 334-347.

Moranz, R., Debinski, D., Winkler, L., Trager, J., McGranahan, D., Engle, D., and Miller, J. 2013. Effects of grassland management practices on ant functional groups in central North America. *Journal of Insect Conservation* 17: 699-713.

Moretti, M., Conedera, M., Duelli, P., and Edwards, P. J. 2002. The effects of wildfire on ground-active spiders in deciduous forests on the Swiss southern slope of the Alps. *Journal of Applied Ecology* 39: 321–336.

Moretti, M., and Legg, C. 2009. Combining plant and animal traits to assess community functional responses to disturbance. *Ecography* 32: 299–309.

Moretti, M., Obrist, M. K., and Duelli, P. 2004. Arthropod biodiversity after forest fires: winners and losers in the winter fire regime of the southern Alps. *Ecography* 27: 173–186.

Munro, N. T., Kovac, K., Niejalke, D., and Cunningham, R. B. 2009. The effect of a single burn event on the aquatic invertebrates in artesian springs. *Austral Ecology* 34: 837–847.

Murdoch, W. W., Evans, F. C., and Peterson, C. H. 1972. Diversity and pattern in plants and insects. *Ecology* 53: 819-829.

Nakamura, A., Catterall, C. P., House, A. P. N., Kitching, R. L., and Burwell, C. J. 2007. The use of ants and other soil and litter arthropods as bio-indicators of the impacts of rainforest clearing and subsequent land use. *Journal of Insect Conservation* 11: 177-186.

Neeson, T. M., Van Rijn, I., and Mandelik, Y. 2013. How taxonomic diversity, community structure, and sample size determine the reliability of higher taxon surrogates. *Ecological Applications* 23: 1216–1225.

Nelson, J. M. 1965. A seasonal study of aerial insects close to a moorland stream. *Journal of Animal Ecology* 34: 573–579.

Neumann, F. G., and Tolhurst, K. 1991. Effects of fuel reduction burning on epigeal arthropods and earthworms in dry sclerophyll forest of west-central Victoria. *Australian Journal of Ecology* 16: 315–330.

New, T. R. 1979. Phenology and relative abundance of Coleoptera on some Australian Acacias. *Australian Journal of Zoology* 27: 9-16.

New, T. R. 2014. *Insects, Fire and Conservation*. Springer, Cham, Switzerland.

Nicolls, K. D., and Dimmock, G. M. 1965. Soils. Pages 26–29 *in* J. L. Davies, editor. *Atlas of Tasmania*. Mercury Press, Hobart.

Nielsen, B. O. 1974. The phenology of beech canopy insects in Denmark. *Videnskabelige Meddelelser fra Dansk Naturhistorisk Forening* 137: 95–124.

Niwa, C. G., and Peck, R. W. 2002. Influence of prescribed fire on carabid beetle (Carabidae) and spider (Araneae) assemblages in forest litter in southwestern Oregan. *Environmental Entomology* 31: 785–796.

Noble, I. R., and Slatyer, R. O. 1980. The use of vital attributes to predict successional changes in plant communities subject to recurrent disturbances. *Vegetatio* 43: 5–21.

Noss, R. F. 1990. Indicators for monitoring biodiversity - a hierarchical approach. *Conservation Biology* 4: 355-364.

Novotny, V., and Basset, Y. 1998. Seasonality of sap-sucking insects (Auchenorrhyncha, Hemiptera) feeding on *Ficus* (Moraceae) in a lowland rain forest in New Guinea. *Oecologia* 115: 514–522.

O'Dowd, D. J., and Gill, A. M. 1984a. Predator Satiation and Site Alteration Following Fire: Mass Reproduction of Alpine Ash (Eucalyptus Delegatensis) in Southeastern Australia. *Ecology* 65: 1052-1066.

O'Dowd, D. J., and Gill, A. M. 1984b. Predator satiation and site alteration: mass reproduction of Alpine Ash (*Eucalyptus delegatensis*) following fire in southeastern Australia. *Ecology* 65: 1052–1066.

Ohmart, C. P., Stewart, L. G., and Thomas, J. R. 1983. Phytophagous insect communities in the canopies of three Eucalyptus forest types in south-eastern Australia. *Australian Journal of Ecology* 8: 395-403.

Olsgard, F., Somerfield, P. J., and Carr, M. R. 1997. Relationships between taxonomic resolution and data transformations in analyses of a macrobentic community along an established pollution gradient. *Marine Ecology Progress Series* 149: 173–181.

Olsgard, F., Somerfield, P. J., and Carr, M. R. 1998. relationships between taxonomic resolution, macrobenthic community patterns and disturbance. *Marine Ecology Progress Series* 172: 25–36.

Palmer, C. 2010. Chronological changes in terrestrial insect assemblages in the arid zone of Australia. *Environmental Entomology* 39: 1775–1787.

Panzer, R. 2002. Compatibility of presecribed burning with the conservation of insects in small, isolated prairie reserves. *Conservation Biology* 16: 1296–1307.

Parr, C. L., and Chown, S. L. 2003. Burning issues for conservation: A crtique of faunal fire research in Southern Africa. *Austral Ecology* 28: 384–395.

Parr, C. L., Robertson, H. G., Biggs, H. C., and Chown, S. L. 2004. Response of African savanna ants to long-term fire regimes. *Journal of Applied Ecology* 41: 630–642.

Pausas, J. G., Keeley, J. E., and Verdú, M. 2006. Inferring differing evolutionary processes of plant persistence traits in northern hemisphere Mediterranean fire-prone ecosystems. *Journal of Ecology* 94: 31–30.

Pinheiro, F., Diniz, I. R., Coelho, D., and Bandeira, P. S. 2002. Seasonal pattern of insect abundance in the Brazilian cerrado. *Austral Ecology* 27: 132–136.

Prestidge, R. A. 1982. The influence of nitrogenous fertilizer on the grassland Auchenorrhyncha (Homoptera). *Journal of Applied Ecology* 19: 735-749.

Price, P. W., Denno, R. F., Eubanks, M. D., Finke, D. L., and Kaplan, I. 2011. *Insect Ecology: behaviour, Populations and Communities*. Cambridge, Cambridge University Press.

Pyke, G. H. 1983. Seasonal pattern of abundance of honeyeaters and their resources in heathland areas near Sydney. *Australian Journal of Ecology* 8: 217–233.

Pyrke, J. S., and Samways, M. J. 2012a. Differential resilience of invertebrates to fire. *Austral Ecology* 37: 460–469.

Pyrke, J. S., and Samways, M. J. 2012b. Importance of using many taxa and having adequate controls for monitoring impacts of fire for arthropod conservation. *Journal of Insect Conservation* 16: 177–185.

Recher, H. F., Gowing, G., Kavanagh, R., Shields, J., and Rohan-Jones, W. 1983. Birds, resources and time in a tablelands forest. *Proceedings of the Ecological Society of Australia* 12: 101–123.

Recher, H. F., Majer, J. D., and Ganesh, S. 1996. Seasonality of canopy invertebrate communities in eucalypt forests of eastern and western Australia. *Australian Journal of Ecology* 21: 64–80.

Richardson, M. L., and Hanks, L. M. 2009. Effects of Grassland Succession on Communities of Orb-Weaving Spiders. *Environmental Entomology* 38: 1595-1599.

Rosser, N., and Eggleton, P. 2012. Can higher taxa be used as a surrogate for species-level data in biodiversity surveys of litter/soil insects? *Journal of Insect Conservation* 16: 87-92.

Ruston, S. P., Luff, M. L., and Eyre, M. D. 1991. Habitat characteristics of grassland *Pterostichus* species (Coleoptera, Carabidae). *Ecological Entomology* 16: 91–104.

Sanderson, R. A., Rushton, S. P., Cherrill, A. J., and Byrne, J. P. 1995. Soil, vegetation and space: an analysis of their effects on the invertebrate communities of amoorland in north-east England. *Journal of Applied Ecology* 32: 506–518.

Santos, E. M. R., Franklin, E., and Luizão, F. J. 2008. Litter manipulation and associated invertebrate fauna in secondary forest, central Amazonia, Brazil. *Acta Oecologica* 34: 274–284.

Schaffers, A. P., Raemakers, I. P., Sykora, K. V., and Ter Braak, C. J. F. 2008. Arthropod assemblages are best predicted by plant species composition. *Ecology* 89: 782-794.

Schipper, A. M., Lotterman, K., Geertsma, M., Leuven, R., and Hendriks, A. J. 2010. Using datasets of different taxonomic detail to assess the influence of floodplain characteristics on terrestrial arthropod assemblages. *Biodiversity and Conservation* 19: 2087-2110.

Sedlacek, J. D., Barrett, G. W., and Shaw, D. R. 1988. Effects of nutrient enrichment on the Auchenorrhyncha (Homoptera) in contrasting grassland communites. *Journal of Applied Ecology* 25: 537-550.

Sieman, E., Haarstad, J., and Tilman, D. 1997. Short-term and long-term effects of burning on oak savanna arthropods. *American Midland Naturalist* 137: 349–361.

Siemann, E., Tilman, D., Haarstad, J., and Ritchie, M. E. 1998. Experimental tests of the dependence of arthropod diversity on plant diversity. *American Naturalist* 152: 738–750.

Somerfield, P. J., and Clarke, K. R. 1995. Taxonomic levels, in marine community studies revisited. *Marine Ecology Progress Series* 127: 113–119.

Southwood, T. R. E., Brown, V. K., and Reader, P. M. 1979. The relationships of plant and insect diversities in succession. *Biological Journal of the Linnean Society* 12: 327–348.

Southwood, T. R. E., Wint, G. R. W., Kennedy, C. E. J., and Greenwood, S. R. 2004. Seasonality, abundance, species richness and specificity of the phytophagus guild of insects on oak (*Quercus*) canopies. *European Journal of Entomology* 101: 43–50.

Springett, J. A. 1976. The effect of prescribed burning on the soil fauna and on litter decomposition in Western Australian forests. *Australian Journal of Ecology* 1: 77–82.

Storey, K. 2010. A review of the potential interations between fire, soil, hydrology and geomorphology of buttongrass moorland. Pages 10–21 *in* J. Balmer, editor. *Proceedings of the 2007 Buttongrass Moorland Management Workshop*. Department of Primary Industries, Parks, Water and Environment, Hobart Tasmania.

Stork, N. E., and Eggleton, P. 1992. Invertebrates as determinants and indicators of soil quality. *American Journal of Alternative Agriculture* 7: 38–47.

Swengel, A. B. 1996. Effects of fire and hay management on abundance of prairie butterflies. *Biological Conservation* 76: 73-85.

Swengel, A. B. 2001. A literature review of insect responses to fire, compared to other conservation management of open habitat. *Biodiversity and Conservation* 10: 1141–1169.

Symonds, M. R. E., and Moussalli, A. 2011. A brief guide to model selection, multimodel inference and model averaging in behavioural ecology using Akaike's information criterion. *Behavioural Ecology and Sociobiology* 65: 13–21.

Taylor, J. E., and Fox, B. J. 2001. Disturbance effects from fire and mining produce different lizard communities in eastern Australian forests. *Austral Ecology* 26: 193-204.

Teasdale, L. C., Smith, A. L., Thomas, M., Whitehead, C. A., and Driscoll, D. A. 2013. Detecting invertebrate responses to fire depends on sampling method and taxonomic resolution. *Austral Ecology* 38: 874-883.

Thomas, I. 1993. Late Pleistocene environments and aboriginal settlement patterns in Tasmania. *Australian Archaeology* 36: 1–11.

Timms, L. L., Bowden, J. J., Summerville, K. S., and Buddle, C. M. 2013. Does species-level resolution matter? Taxonomic sufficiency in terrestrial arthropod biodiversity studies. *Insect Conservation and Diversity* 6: 453–462.

Tinner, W., Conedera, M., Ammann, B., and Lotter, A. F. 2005. Fire ecology north and south of the Alps since the last ice age. *The Holocene* 15: 1214-1226.

Tscharntke, T., and Greiler, H. J. 1995. Insect communities, grasses, and grasslands. *Annual Review of Entomology* 40: 535–558.

TVMP. 2004. *Tasmanian Vegetation Mapping Program. Western Tasmanian Wilderness World Heritage Area Vegetation Mapping*. Department of Primary Industries, Parks, Water and Environment, Tasmania, Hobart.

Underwood, E. C., and Christian, C. E. 2009. Consequences of prescribed fire and grazing on grassland ant communities. *Environmental Entomology* 38: 325–332.

Underwood, E. C., and Quinn, J. F. 2010. Response of ants and spiders to prescribed fire in oak woodlands of California. *Journal of Insect Conservation* 14: 359–366.

van Heurck, P., and Abbot, I. 2003. Fire and terrestrial invertebrates in south-west Western Australia. Pages 291–319 *in* I. Abbott and N. Burrows, editors. *Fire in Ecosystems of South-west Western Australia: Impacts and Management*. Backhuys Publishers, Leiden, The Netherlands.

van Rijn, I., Neeson, T. M., and Mandelik, Y. 2015. Reliability and refinement of the higher taxa approach for bee richness and composition assessments. *Ecological Applications* 25: 88-98.

Vieira, L. C., Oliveira, N. G., Brewster, C. C., and Gayubo, S. F. 2012. Using higher taxa as surrogates of species-level data in three Portuguese protected areas: a case study on Spheciformes (Hymenoptera). *Biodiversity and Conservation* 21: 3467-3486.

von Platen, J., Kirkpatrick, J. B., and Allen, K. J. 2011. Variation in decadal fire frequency in southeastern Tasmanian dry eucalypt forest 1740-2004. *Australian Forestry* 74: 180–189.

Walker, L. R., Wardle, D. A., Bardgett, R. D., and Clarkson, B. D. 2010. The use of chronosequences in studies of ecological succession and soil development. *Journal of Ecology* 98: 725–736.

Wallner, A. M., Molano-Flores, B., and Dietrich, C. H. 2012. The influence of fire on Illinois hill praire Auchenorrhyncha (Insecta: Hemiptera) diversity and integrity. *Journal of Insect Conservation* 16: 433–445.

Walter, D. E., and Proctor, H. C. 1999. *Mites: ecology, evolution, and behaviour*. University of New South Wales Press, Sydney.

Warren, S. D., Scifres, C. J., and Teel, P. D. 1987. Response of grassland arthropods to burning: a review. *Agriculture Ecosystems and Environment* 19: 105–130.

Weeks, R. D., and Holtzer, T. O. 2000. Habitat and Season in Structuring Ground-Dwelling Spider (Araneae) Communities in a Shortgrass Steppe Ecosystem. *Environmental Entomology* 29: 1164-1172.

Whelan, R. J. 1995. *The Ecology of Fire*. Cambridge University Press, Cambridge.

Whelan, R. J., Rodgerson, L., Dickman, C. R., and Sutherland, E. F. 2002. Critical life cycles of plants and animals: developing a process-based understanding of population changes in fire-prone landscapes. Pages 94–124 *in* R. A. Bradstock, J. E. Williams, and A. M. Gill, editors. *Flammable Australia*. Cambridge University Press, Cambridge.

Willis, E. O. 1976. Seasonal changes in the invertebrate litter fauna of Barro Colorado Island, Panama. *Revista Brasileira de Biologia* 36: 643–657.

Wills, T. J. 2003. Using *Banksia* (Proteaceae) node counts to estimate time since fire. *Australian Journal of Botany* 51: 239-242.

Winqvist, C., Bengtsson, J., Öckinger, E., Aavik, T., Berendse, F., Clement, L., Fischer, C., Flohre, A., Geiger, F., Liira, J., Thies, C., Tscharntke, T., Weisser, W., and

Bommarco, R. 2014. Species' traits influence ground beetle responses to farm and landscape level agricultural intensification in Europe. *Journal of Insect Conservation* 18: 837-846.

Woinarski, J. C. Z., and Cullen, J. M. 1984. Distribution of invertebrates on foliage in forests of southeastern Australia. *Australian Journal of Ecology* 9: 207-232.

Wolda, H. 1979. Seasonality parameters for insect populations. *Research on Population Ecology* 20: 247–256.

Wolda, H. 1988. Insect Seasonality: Why? *Annual Review of Ecological Systems* 19: 1–18.

Wolda, H., and Chandler, D. S. 1996. Diversity and seasonality of tropical Pselaphidae and Anthicidae (Coleoptera). *Proceedings of the Koninklijke Nederlandse Akademie van Wetenschappen* 99: 313–333.

Woodward, F. I. 1994. How many species are required for a functioning ecosystem? Pages 271–279 *in* E. D. Schultz and H. A. Mooney, editors. *Biodiversity and Ecosystem Function*. Springer-Verlag, Heidelberg.

York, A. 1999a. Long-term effects of frequent low-intensity burning on the abundance of litter-dwelling invertebrates in coastal blackbutt forests of southeastern Australia. *Journal of Insect Conservation* 3: 191–199.

York, A. 1999b. Long-term effects of repeated prescribed burning on forest invertebrates: management implications for the conservation of biodiversity. Pages 181–266 *Australia's Biodiversity: Responses to Fire - Plants, Birds and Invertebrates*. Environment Australia, Canberra.

York, A. 2000. Long-term effects of frequent low-intensity burning on ant communities in coastal blackbutt forests of southeastern Australia. *Austral Ecology* 25: 83–98.

Appendix 1 List of taxa and location of archived specimens

This table lists the taxa identified from surveys for Chapters 2 and 3. Not all taxa could be identified to species or morphospecies. Diptera and Hymenoptera were identified to species or morphospecies in Chapter 2 and to family in Chapter 3. Values are the number of specimens. PF = pifall trap samples, SW = sweep net samples. Specialists primarily responsible for identifying groups of taxa into species or morphospecies are given in parentheses. Several new species have been subsequently formally described and these species are marked with an asterix (*).

Number of taxa identified to species or morphospecies is 1642.
Number of taxa assigned to a described species (includes tentative assignments indicated by “nr” or “?”) is 265.
Number of new taxa recognised is 37 and 11 now have published descriptions.

Most of the collected invertebrates were lodged with the Tasmanian Museum and Art Gallery (TMAG). Araneae and Acarina used in chapters 2 and 3 were lodged with the Queensland Museum. Non-Formicidae Hymenoptera and Tricladida used in chapter 2 were lodged with the Queensland Museum. Acarina voucher specimens were lodged with TMAG. Diptera used in chapters 2 and 3 were lodged with the Australian Museum. Thysanoptera and Trichoptera used in chapters 2 and 3 were lodged with the Australian Insect Collection. Hemiptera used in chapter 2 were lodged with the Australian Museum and the Hemiptera voucher specimens were lodged with TMAG. Chilopoda, Diplopoda and some Formicidae and Isopoda were lodged with the Queen Victoria Museum and Art Gallery. Formicidae voucher specimens were lodged with TMAG. Coleoptera voucher specimens were lodged in Peter McQuillan’s collection at the University of Tasmania.

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
ANNELIDA										
OLIGOCHAETA (Geoff Dyne)										
Haplotaxida										
Megascolecidae	*Perionychella bithecata* (Jamieson, 1974)	0	0	1	0	1	0	0	1	1
	Perionychella sp. 1	0	0	1	0	1	0	0	1	1
Total Oligochaeta		0	0	2	0	2	0	0	2	2
ARTHROPODA										
ARACHNIDA										
Acarina (Owen Seeman)										
Endeostigmata										
Bimichaelidae	*Bimichaela* sp. 1	7	6	19	3	26	9	13	22	35
	Bimichaelidae sp. 1	0	0	7	0	7	0	0	7	7
	Family indet.	0	0	4	1	4	1	0	5	5
	Endeostigmata sp. 1	0	0	4	0	4	0	0	4	4
	Endeostogmata sp. 1	0	0	0	1	0	1	0	1	1
Nanorchestidae	*Nanorchestes* sp. 1	0	0	10	0	1	0	0	10	10
Ixodida										
Ixodidae	*Ixodes trichosuri* Roberts, 1960	0	1	0	0	0	1	1	0	1
Mesostigmata										
Ascidae	*Amblyseius* sp. 1	3	0	0	0	3	0	3	0	3
	Asca foliata Womersley, 1956	0	0	1	0	1	0	0	1	1
	Asca porosa Wood, 1966	0	0	1	0	1	0	0	1	1
	Asca sp. nov. 1	0	0	0	1	0	1	0	1	1
	Asca sp. nov. 2	0	34	0	7	0	41	34	7	41
	Ascidae sp. 1	1	0	0	0	1	0	1	0	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Cheiroseius sp. nov.	2	0	1	0	3	0	2	1	3
	Gamasellodes nr *bicolor* (Berlese, 1918)	0	0	3	0	3	0	0	3	3
	Lasioseius boomsmai Womersley, 1956	1	0	0	0	1	0	1	0	1
	Lasioseius sp. 1	0	0	1	0	1	0	0	1	1
Halolaelapidae	*Halolaelaps* sp. nov.	0	0	1	0	1	0	0	1	1
Laelapidae	*Geolaelaps* sp. 1	0	0	1	0	1	0	0	1	1
	Hypoaspis sp. 1	0	0	9	0	9	0	0	9	9
	Laelapidae sp. 1	0	0	1	0	1	0	0	1	1
	Laelapidae sp. 2	1	0	1	0	2	0	1	1	2
	Laelapidae sp. 3	1	0	0	0	1	0	1	0	1
	Pseudoparasitus australicus (Womersley, 1956)	1	0	14	0	15	0	1	14	15
Macrochelidae	*Macrocheles mykytowyczi* Womersley, 1956	3	0	13	0	16	0	3	13	16
Ologamasidae	*Acugamasus* sp. 1	6	0	0	0	6	0	6	0	6
	Caliphis sp. nov.	0	0	2	0	2	0	0	2	2
	Euepicrius sp. 1	5	0	0	0	5	0	5	0	5
	Euepicrius sp. 2	0	0	8	0	8	0	0	8	8
	Euepicrius sp. nov. 1	0	0	3	0	3	0	0	3	3
	Gamasellus sp. 1	2	0	1	0	3	0	2	1	3
	Gamasiphis sp. 1	1	0	0	0	1	0	1	0	1
	Gamasiphis sp. 2	0	0	2	0	2	0	0	2	2
	Gamasiphis sp. 3	0	0	3	0	3	0	0	3	3
	Gamasiphis sp. 4	0	0	2	0	2	0	0	2	2
	Ologamasidae sp. 1	0	1	0	0	0	1	1	0	1
	Ologamasidae sp. 2	0	0	1	0	1	0	0	1	1
Phytoseiidae	*Australiseiulus* nr *angophorae* (Schicha, 1981)	0	1	0	0	0	1	1	0	1
	Phytoseiidae sp. 1	0	1	0	0	0	1	1	0	1
	Typhlodromus singularis Chant, 1957 group	0	3	0	6	0	9	3	6	9
Rhodacaridae	*Acugamasus* sp. nov.	1	0	4	0	5	0	1	4	5
	Athiasella dentata (Womersley, 1942)	0	0	1	0	1	0	0	1	1
	Rhodacaridae sp. 1	0	0	4	0	4	0	0	4	4

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Rhodacaridae sp. 2	1	0	2	1	3	1	1	3	4
	Rhodacaridae?	0	0	4	0	4	0	0	4	4
	Rhodacaridae? sp. 3	0	0	4	0	4	0	0	4	4
Uropodidae	Uropodidae sp. 1	0	0	1	0	1	0	0	1	1
	Uropodidae sp. 2	0	0	1	0	1	0	0	1	1
	Uropodidae sp. 3	1	0	0	0	1	0	1	0	1
	Uropodidae sp. 4	0	0	3	0	3	0	0	3	3
Mesostigmata indet.	Mesostigmata sp. 1	0	0	1	0	1	0	0	1	1
	Mesostigmata sp. 2	0	0	2	1	2	1	0	3	3
Oribatida										
Cepheidae	Cepheidae sp. 1	0	0	5	0	5	0	0	5	5
Crassoribatulidae	*Fovoribatula brevisetosa* Lee & Birchby, 1991	1	9	4	287	5	296	10	291	301
	Fovoribatula mesosetosa Lee & Birchby, 1991	0	0	0	2	0	2	0	2	2
Cymbaeremaeidae	*Scapheremaeus* sp. 1	2	62	0	0	2	62	64	0	64
Haplozetidae	*Magnobates* sp. nov.	0	1	0	0	0	1	1	0	1
Hydrozetidae	*Hydrozetes lemnae* (Coggi, 1897)	0	0	1	0	1	0	0	1	1
Liebstadiidae	*Maculobates endroedyyoungai* Mahunka 1989	3	1	0	0	3	1	4	0	4
	Reductobates humeratus Balogh & Mahunka, 1966	6	0	67	99	73	99	6	166	172
Mochlozetidae	Mochlozetidae sp. 1	0	0	14	0	14	0	0	14	14
Mycobatidae	*Antarctozetes* nr. *longicaulis* (Hammer, 1967)	2	0	114	46	116	46	2	160	162
	Antarctozetes sp. 1	0	0	0	16	0	16	0	16	16
	Baloghobates nr. *nudus* Hammer, 1967	0	0	1	1	1	1	0	2	2
	Baloghobates sp. nov. 1	12	816	0	5	12	821	828	5	833
	Baloghobates sp. nov. 2	0	1924	0	42	0	1966	1924	42	1966
	Baloghobates sp. nov. 3	94	0	105	0	199	0	94	105	199
	Mycobatidae sp. 1	0	0	1	0	1	0	0	1	1
	Neomycobates sp. 1	2	0	1	0	3	0	2	1	3
	Neomycobates sp. 2	1	0	0	0	1	0	1	0	1
Nanhermanniidae	*Nanhermannia domrowi* Balogh & Mahunka, 1978	0	0	1	0	1	0	0	1	1
Neotrichozetidae	*Neotrichozetes* sp. 2	0	1	0	0	0	1	1	0	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Neotrichozetes sp. 3	3	0	0	0	3	0	3	0	3
	Neotrichozetes sp. nov.	0	0	1	0	1	0	0	1	1
Oppiidae	*Lanceoppia* nr. *microtricha* Balough & Mahunka, 1975	1	0	0	0	1	0	1	0	1
	Lanceoppia pertineata Mahunka, 1989	4	0	0	0	4	0	4	0	4
	Lanceoppia sp. nov.	0	0	0	1	0	1	0	1	1
	Lanceoppia tasmanica Mahunka, 1989	1	0	4	0	5	0	1	4	5
Oribatellidae	*Safrobates miniporus* Mahunka, 1989	3	0	141	2	144	2	3	143	146
Oribatulidae	*Maculobates* sp. 1	22	3	62	17	84	2	25	79	104
	Maculobates sp. 3	3	0	3	0	6	0	3	3	6
	Maculobates sp. 4	1	0	1	0	2	0	1	1	2
	Oribatulidae sp. 1	0	0	1	0	1	0	0	1	1
	Paraphauloppia nr. *globata* Lee & Birchby, 1991	0	0	6	0	6	0	0	6	6
	Zygooribatula cycloporosa Lee, 1992	2	41	6	70	8	111	43	76	119
Otocepheidae	*Pseudotocepheus* sp. nov. 1	0	3	0	0	0	3	3	0	3
	Pseudotocepheus sp. nov. 2	0	4	0	0	0	4	4	0	4
Parakalummatidae	*Sandenia rotunda* Wallwork, 1966	18	2210	76	149	94	2359	2228	225	2453
Peloppiidae	nr *Pseudoceratoppia* sp.	0	0	1	0	1	0	0	1	1
Steganacaridae	*Notophthiracarus* sp. 1	1	0	0	0	1	0	1	0	1
Oribatida indet.	Oribatida sp. 1	0	0	2	0	2	0	0	2	2
	Oribatida sp. 2	0	0	2	0	2	0	0	2	2
	Oribatida sp. 3	0	0	1	0	1	0	0	1	1
	Oribatida sp. 4	0	0	1	0	1	0	0	1	1
	Oribatida sp. 5	0	1	0	0	0	1	1	0	1
	Oribatida sp. 6	0	3	0	0	0	3	3	0	3
	Oribatida sp. 7	0	0	0	8	0	8	0	8	8
	Oribatida sp. 8	8	1120	1	45	9	1165	1128	46	1174
	Oribatida sp. 9	0	0	1	27	1	27	0	28	28
	Oribatida sp. 10	0	0	9	21	9	21	0	30	30
	Oribatida sp. 11	0	0	0	1	0	1	0	1	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Oribatida sp. 12	1	0	0	0	1	0	1	0	1
	Oribatida sp. 13	0	3	0	0	0	3	3	0	3
	Oribatida sp. 14	0	2	0	0	0	2	2	0	2
Prostigmata										
Adamystidae	*Adamystis* sp. nov.	1	1	1	0	2	1	2	1	3
Anystidae	*Chaussieria* sp. 1	0	0	26	3	26	3	0	29	29
	Walzia australica Womersley, 1942	20	484	111	1098	131	1582	504	1209	1713
Bdellidae	*Bdellodes currax* Atyeo, 1963	0	4	4	1	4	5	4	5	9
	Bdellodes harpax Atyeo, 1963	97	0	58	2	155	2	97	60	157
	Bdellodes hickmani (Womersley, 1933)	2	0	7	0	9	0	2	7	9
	Bdellodes spp.	25	16	52	13	77	29	41	65	106
Calyptostomatidae	Calyptostomatidae sp. 1	1	1	0	0	1	1	2	0	2
Cheyletidae	Cheyletidae sp. 1	2	0	0	0	2	0	2	0	2
Chyzeriidae	*Chyzeria* sp. 1	2	0	0	0	2	0	2	0	2
	Chyzeriidae sp. 1	0	2	0	2	0	4	2	2	4
Cunaxidae	*Armascirus* sp. 1	7	0	0	1	7	1	7	1	8
	Cunaxidae sp. 1	0	0	1	1	1	1	0	2	2
Eriorhynchidae	*Eriorhynchus* nr. *hades* Qin & Halliday, 1997	0	0	1	0	1	0	0	1	1
	Eriorhynchus ramosus Qin & Halliday, 1997	95	0	105	2	2	2	95	107	202
Erythraeidae	*Abrolophus* sp. 1	1	2	1	0	2	2	3	1	4
	Caeculisoma sp. 1	0	0	2	0	2	0	0	2	2
	Curteria sp. 1	7	0	22	0	29	0	7	22	29
	Curteria sp. 2	0	0	3	0	3	0	0	3	3
	Curteria sp. 3	0	0	1	0	1	0	0	1	1
	Grandjeanella sp. nov.	0	2	0	0	0	2	2	0	2
	Leptus nr *agrotis* Southcott, 1993	19	14	83	4	12	18	33	87	120
	Leptus nr charon Southcott, 1991	47	9	55	8	12	17	56	63	119
	Leptus truncatus Southcott, 1993	0	3	24	15	24	18	3	39	42
	Leptus sp. 1	14	0	20	2	34	2	14	22	36
	Leptus sp. 2	0	0	6	0	6	0	0	6	6

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Leptus sp. 3	0	0	1	0	1	0	0	1	1
	Microsmaris sp. 1	0	0	13	26	13	26	0	39	39
	Mypongia sp. nov.	0	4	0	7	0	11	4	7	11
	Rainbowia sp. 1	0	0	0	3	0	3	0	3	3
	Rainbowia sp. 2	3	0	13	0	16	0	3	13	16
	Wartookia rebeccae Southcott, 1961	4	67	13	97	17	164	71	110	181
	Wartookia sp. nov. 1	5	1120	7	1075	12	2195	1125	1082	2207
	Wartookia sp. nov. 2	0	34	2	120	2	154	34	122	156
	Wartookia sp. nov. 3	0	2	3	0	3	2	2	3	5
Eupodidae	*Eupodes* sp. 1	6	0	29	0	35	0	6	29	35
	nr *Eupodes* sp. nov.	0	0	256	2	256	2	0	258	258
Halacaridae	Halacaridae sp. 1	2	0	0	0	2	0	2	0	2
Johnstonianidae	*Centrothrombium* sp. 1	9	0	7	0	16	0	9	7	16
	Crossothrombium sp. 1	0	0	2	0	2	0	0	2	2
	Diplothrombium sp. 1	2	0	0	2	2	2	2	2	4
	Johnstonianidae sp. 1	3	2	0	0	3	2	5	0	5
	Paraplothrombium sp. 1	3	0	0	0	3	0	3	0	3
Microtrombidiidae	*Baundikia* sp. 1	4	0	9	3	13	3	4	12	16
	Ctnerythraeus sp. 1	7		3		1		7	3	10
	Microtrombididae sp. 1			1	1	1	1		2	2
	Workandella virgata Southcott, 1994			2		2			2	2
	Platytrombidium sp. 1	1		1		2		1	1	2
	Tropicotrombium sp. 1	3		3		6		3	3	6
Neotrombidiidae	Neothrombiidae sp. 1			2		2			2	2
	Acritrombium sp. 1				1		1		1	1
Penthaleidae	*Halotydeus* sp. 1	107		130	1	237	1	107	131	238
Penthalodidae	*Stereotydeus pseudopulcher* Qin, 1994	3		15		18		3	15	18
	Stereotydeus tasmanicus Qin, 1994	9				9		9		9
Pyemotidae	Pyemotidae sp. 1				2		2		2	2
Rhagidiidae	*Rhagidia* sp. 1	9		16		25		9	16	25

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Rhagidiidae sp. 1			8	3	8	3		11	11
Smarididae	*Hirstiosoma* sp. 1	20		107		127		20	107	127
	Hirstiosoma sp. 2			2		2			2	2
	Hirstiosoma tasmaniensis Womersley & Southcott, 1941	1		57	1	58	1	1	58	59
Stigmaeidae	*Cheylostigmaeus* sp. 1	2				2		2		2
	Eustigmaeus sp. nov.	1				1		1		1
	Stigmaeidae sp. 1			1		1			1	1
Tarsonemidae	*Tarsonemus* sp. 1		7				7	7		7
Tetranychidae	*Monoceronychus* sp. 1				1		1		1	1
	Tetranychus nr. *lambi* Ptrichard & Baker, 1955			3	29	3	29		32	32
Trombellidae	Trombellidae sp. 1				1		1		1	1
Trombiculidae	Trombiculidae sp. 1			1		1			1	1
Trombidiidae	*Allothrombium* sp. 1		10		1		11	10	1	11
Tydeidae	*Australotydeus* sp. 1				2		2		2	2
	Pseudotydeus sp. 1			14		14			14	14
	Tydeidae sp. 1			5		5			5	5
Prostigmata indet.	Prostigmata sp. 1				3		3		3	3
	Prostigmata sp. 2	1	0	0	0	1	0	1	0	1
Total Acarina		773	8035	2082	3393	2459	11410	8808	5475	14283
Araneae (Robert Raven)										
Agelenidae	Agelenidae sp. 1	0	220	0	54	0	274	220	54	274
Amaurobiidae	Amaurobiidae sp. 1	0	0	2	0	2	0	0	2	2
	Amaurobiidae sp. 2	3	1	6	0	9	1	4	6	10
	Amaurobiidae sp. 3	0	1	0	2	0	3	1	2	3
	Amaurobiidae sp. 4	2	0	1	3	3	3	2	4	6
	Amaurobiidae sp. 5	0	0	4	0	4	0	0	4	4
	Amaurobiidae sp. 6	24	0	30	3	54	3	24	33	57
	Amaurobiidae sp. 7	46	0	32	0	78	0	46	32	78

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Amaurobiidae sp. 8	111	0	70	0	181	0	111	70	181
	Amaurobiidae sp. 9	16	0	11	0	27	0	16	11	27
	Amaurobiidae sp. 10	1	0	0	0	1	0	1	0	1
	Amaurobiidae sp. 11	1	0	2	0	3	0	1	2	3
	Amaurobiidae sp. 12	3	0	5	0	8	0	3	5	8
	Manjala sp. 1	2	1	9	0	11	1	3	9	12
	Manjala sp. 2	0	0	2	0	2	0	0	2	2
	Otira sp. 1	1	0	25	0	26	0	1	25	26
Amphinectidae	**Tanganoides mcpartlan* Davies, 2003	4	0	0	0	4	0	4	0	4
	Tasmarubrius hickmani Davies, 1998	1	0	170	2	171	2	1	172	173
	Tasmarubrius truncus Davies, 1998	15	0	106	0	121	0	15	106	121
	Tasmarubrius sp. 1	0	0	1	0	1	0	0	1	1
	Tasmarubrius sp. 2	0	0	6	0	6	0	0	6	6
	cf. *Tasmarubrius*sp. 3	0	0	1	0	1	0	0	1	1
	**Teeatta driesseni* Davies, 2005	0	0	1	0	1	0	0	1	1
Anapidae	Anapidae sp. 1	0	0	1	0	1	0	0	1	1
	Anapidae sp. 2	2	0	0	0	2	0	2	0	2
	Hickmanapis sp.1	0	0	1	1	1	1	0	2	2
Araneidae	Araneidae sp. 1	0	1	0	11	0	12	1	11	12
	Araneus acuminatus (L. Koch, 1872)	0	6	0	370	0	376	6	370	376
	Araneus arenaceus (Keyserling, 1886)	1	67	1	37	2	14	68	38	106
	Araneus bradleyi (Keyserling, 1887)	0	10	0	17	0	27	10	17	27
	*Araneus brisbanae (*L. Koch, 1867)	0	1	0	1	0	2	1	1	2
	Araneus eburnus (Keyserling, 1886)	0	8	0	9	0	17	8	9	17
	Araneus lodicula (Keyserling, 1887)	0	0	0	13	0	13	0	13	13
	Araneus sp. 1	0	5	1	3	1	8	5	4	9
	Araneus sp. 2	0	52	0	115	0	167	52	115	167
	Araneus sp. 3	0	35	1	25	1	6	35	26	61
	Araneus sp. 4	0	0	0	4	0	4	0	4	4
	Araneus sp. 5	0	9	0	28	0	37	9	28	37

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Araneus sp. 6	0	11	0	11	0	22	11	11	22
	Araneus sp. 7	0	3	0	17	0	2	3	17	20
	Araneus sp. 8	0	0	0	4	0	4	0	4	4
	Araneus sp. 9	0	22	0	14	0	36	22	14	36
	Araneus sp. 10	0	3	0	0	0	3	3	0	3
	Araneus sp. 11	0	3	0	1	0	4	3	1	4
	Araneus sp. 12	0	11	0	61	0	72	11	61	72
	Araneus sp. 13	0	0	0	2	0	2	0	2	2
	Araneus sp. 14	0	0	0	1	0	1	0	1	1
	Araneus sp. 15	0	5	0	2	0	7	5	2	7
	Araneus sp. 16	0	2	0	3	0	5	2	3	5
	Araneus sp. 17	0	0	0	1	0	1	0	1	1
	Araneus sp. 18	0	2	0	4	0	6	2	4	6
	Araneus sp. 19	0	1	0	42	0	43	1	42	43
	Araneus sp. 20	0	1	0	0	0	1	1	0	1
	Araneus sp. 21	0	1	0	0	0	1	1	0	1
	Araneus sp. 22	0	0	0	1	0	1	0	1	1
	Araneus sp. 23	0	1	0	0	0	1	1	0	1
	Araneus sp. 24	0	2	0	0	0	2	2	0	2
	Araneus sp. 25	0	0	0	1	0	1	0	1	1
	Araneus sp. 26	0	1	0	1	0	2	1	1	2
	Araneus sp. 27	0	1	0	0	0	1	1	0	1
	Archemorus sp. 1	0	2	0	1	0	3	2	1	3
	Arkys sp. 1	0	1	0	0	0	1	1	0	1
	Cyclosa sp. 1	0	53	0	6	0	59	53	6	59
	Dolophones sp. 1	0	6	0	9	0	15	6	9	15
	Eriophora pustulosa (Walckenear, 1842)	0	38	1	566	1	64	38	567	605
	Eriophora sp. 1	0	3	0	0	0	3	3	0	3
	Eriophora sp. 2	0	77	1	4	1	81	77	5	82
	Eriophora sp. 3	0	1	0	0	0	1	1	0	1
	cf. *Gea* sp. 1	0	0	0	2	0	2	0	2	2

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Larinia sp. 1	0	0	0	1	0	1	0	1	1
	Neoscona sp. 1	0	7	0	44	0	51	7	44	51
Clubionidae	*Cheiracanthium* sp. 1	0	49	1	7	1	56	49	8	57
	Clubiona sp. 1	2	7	0	29	2	36	9	29	38
	Clubiona sp. 2	1	0	0	0	1	0	1	0	1
	Clubiona sp. 3	1	5	0	0	1	5	6	0	6
	Clubiona sp. 4	0	11	0	50	0	61	11	50	61
	Clubiona sp. 5	0	0	0	1	0	1	0	1	1
	Clubiona sp. 6	0	0	2	0	2	0	0	2	2
	Clubiona sp. 7	0	5	2	3	2	8	5	5	10
	Clubiona sp. 8	0	33	3	134	3	167	33	137	170
	Clubiona sp. 9	0	1	0	0	0	1	1	0	1
	Clubiona sp. 10	1	0	0	0	1	0	1	0	1
Corinnidae	*Poecilipta* sp. 1	2	0	2	0	4	0	2	2	4
	*Supunna picta (*L. Koch, 1873)	1	0	0	0	1	0	1	0	1
	Supunna versicolor Simon, 1896	0	0	1	0	1	0	0	1	1
Cyatholipidae	*Matilda* sp. 1	1	0	0	0	1	0	1	0	1
Cycloctenidae	*Cycloctenus* sp. 1	2	0	4	0	6	0	2	4	6
	Cycloctenus sp. 2	3	2	0	0	3	2	5	0	5
	Cycloctenus sp. 3	2	0	0	0	2	0	2	0	2
Desidae	*Badumna* nr *socialis* (Rainbow, 1905)	0	0	0	1	0	1	0	1	1
	Badumna sp. 1	0	1	0	12	0	13	1	12	13
	*Phryganoporus vandiemeni (*Gray, 1983)	0	1	0	4	0	5	1	4	5
	Desidae sp. 1	1	0	0	0	1	0	1	0	1
	Forsterina sp. 1	0	0	3	0	3	0	0	3	3
	Laestrygones sp. 1	14	4	21	1	35	5	18	22	40
	Laestrygones sp. 2	0	1	0	1	0	2	1	1	2
	Ommatauxesis macrops Simon, 1903	33	0	5	0	38	0	33	5	38
	Toxops sp. 1	0	0	1	0	1	0	0	1	1
Dictynidae	*Arangina* sp. 1	0	0	2	1	2	1	0	3	3

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Callevopthalmus? sp. 1	0	0	0	1	0	1	0	1	1
	Dictyna sp. 1	0	300	1	289	1	589	300	290	590
	Dictynidae sp. 1	0	101	0	47	0	148	101	47	148
Gnaphosidae	Gnaphosidae sp. 1	0	0	4	0	4	0	0	4	4
	Gnaphosidae sp. 2	3	0	10	0	13	0	3	10	13
	Gnaphosidae sp. 3	0	0	1	0	1	0	0	1	1
	Gnaphosidae sp. 4	14	0	17	0	31	0	14	17	31
Hahniidae	Hahniidae sp. 1	0	1	1	0	1	1	1	1	2
	Scotospilus sp. 1	35	0	1	0	36	0	35	1	36
Lamponidae	*Lampona lomond* Platnick, 2000	2	0	3	0	5	0	2	3	5
Linyphiidae	*Araeoncus* sp. 1	1	1	2	0	3	1	2	2	4
	Diplocephalus cristatus (Blackwall, 1833)	1	0	64	0	65	0	1	64	65
	Erigone prominens Bosenberg & Stand, 1906	2	3	5	3	7	6	5	8	13
	Laperousea sp. 1	1	0	1	0	2	0	1	1	2
	Laperousea sp. 2	12	8	29	4	41	12	20	33	53
	Laperousea sp. 3	1	4	2	11	3	15	5	13	18
	Laperousea sp. 4	0	0	3	1	3	1	0	4	4
	Laperousea sp. 5	20	1	35	2	55	3	21	37	58
	Laperousea sp. 6	1	0	0	1	1	1	1	1	2
	Laperousea sp. 7	0	0	0	2	0	2	0	2	2
	Laperousea sp. 8	0	0	0	1	0	1	0	1	1
	Laperousea sp. 9	0	1	0	0	0	1	1	0	1
	Laperousea sp. 10	0	4	0	18	0	22	4	18	22
	Laperousea sp. 11	0	0	0	1	0	1	0	1	1
	Laperousea sp. 12	0	1	0	1	0	2	1	1	2
	Laperousea sp. 13	7	1	1	1	8	2	8	2	10
	Laperousea sp. 14	10	2	15	1	25	3	12	16	28
	Laperousea sp. 15	1	1	0	0	1	1	2	0	2
	Linyphiidae sp. 1	0	0	1	0	1	0	0	1	1
	Linyphiidae sp. 2	0	0	1	0	1	0	0	1	1
	Linyphiidae sp. 3	1	10	8	35	9	45	11	43	54

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Linyphiidae sp. 4	0	0	1	0	1	0	0	1	1
	Linyphiidae sp. 5	0	0	0	1	0	1	0	1	1
	Linyphiidae sp. 6	0	0	0	1	0	1	0	1	1
	Mynoglenes sp. 1	1	0	0	3	1	3	1	3	4
	Ostearius melanopygus (O.P.-Cambridge, 1879)	0	0	1	0	1	0	0	1	1
Lycosidae	*Lycosa godeffroyi* L. Koch, 1865	19	0	3	0	22	0	19	3	22
	Lycosa sp. 1	0	0	1	0	1	0	0	1	1
	Lycosa sp. 2	19	0	10	0	29	0	19	10	29
	Lycosa sp. 3	5	0	0	0	5	0	5	0	5
	Lycosa sp. 4	3	0	0	0	3	0	3	0	3
	Lycosa sp. 5	0	0	2	0	2	0	0	2	2
	Lycosa sp. 6	6	0	7	0	13	0	6	7	13
	Lycosa sp. 7	0	0	2	0	2	0	0	2	2
	Lycosa sp. 8	7	0	0	0	7	0	7	0	7
	Lycosa sp. 9	8	0	1	0	9	0	8	1	9
	Lycosa sp. 10	0	0	3	0	3	0	0	3	3
	Lycosa sp. 11	1	0	0	0	1	0	1	0	1
	Lycosa sp. 12	1	0	0	0	1	0	1	0	1
	Lycosidae sp. 1	3	0	7	0	1	0	3	7	10
	Lycosidae sp. 2	0	0	1	0	1	0	0	1	1
	Lycosidae sp. 3	1	0	0	0	1	0	1	0	1
	Lycosidae sp. 4	1	0	7	0	8	0	1	7	8
	Lycosidae sp. 5	18	0	13	0	31	0	18	13	31
	Lycosidae sp. 6	3	0	5	0	8	0	3	5	8
	Lycosidae sp. 7	72	0	337	0	49	0	72	337	409
	Lycosidae sp. 8	0	0	3	0	3	0	0	3	3
	Lycosidae sp. 9	5	0	3	0	8	0	5	3	8
	Lycosidae sp. 10	3	0	11	0	14	0	3	11	14
	Lycosidae sp. 11	1	0	1	0	2	0	1	1	2
	Pardosa sp. 1	37	0	44	0	81	0	37	44	81

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Pardosa sp. 2	5	0	1	0	6	0	5	1	6
	Pardosa sp. 3	3	0	0	0	3	0	3	0	3
	Pardosa sp. 4	35	1	152	0	187	1	36	152	188
	Pardosa sp. 5	1	0	0	0	1	0	1	0	1
	Venatrix funesta (C.L. Koch, 1847)	10	0	8	0	18	0	10	8	18
	Venatrix pseudospeciosa Framenau & Vink, 2001	1	0	3	0	4	0	1	3	4
	Venatrix speciosa (L. Koch, 1877)	1	0	0	0	1	0	1	0	1
Micropholcommatidae	Micropholcommatidae sp. 1	3	0	4	1	7	1	3	5	8
	Micropholcommatidae sp. 2	2	0	2	0	4	0	2	2	4
	Micropholcommatidae sp. 3	3	0	0	0	3	0	3	0	3
	Micropholcommatidae sp. 4	6	0	0	0	6	0	6	0	6
	Micropholcommatidae sp. 5	20	3	16	0	36	3	23	16	39
	Textricella sp. 1	7	0	22	4	29	4	7	26	33
	Textricella sp. 2	1	0	0	0	1	0	1	0	1
	Textricella sp. 3	7	1	25	0	32	1	8	25	33
	Textricella sp. 4	0	0	1	0	1	0	0	1	1
	Textricella sp. 5	8	0	17	0	25	0	8	17	25
	Textricella sp. 6	0	0	5	0	5	0	0	5	5
	Textricella sp. 7	1	0	0	0	1	0	1	0	1
Mimetidae	*Australomimetus* sp. 1	1	0	0	2	1	2	1	2	3
	Ero tasmaniensis (Hickman, 1929)	6	0	9	0	15	0	6	9	15
	Mimetidae sp. 1	2	1	5	0	7	1	3	5	8
Miturgidae	*Miturga agelenina* Simon, 1909	3	0	12	0	15	0	3	12	15
	Miturga gilva L. Koch, 1872	1	0	5	0	6	0	1	5	6
Mysmenidae	*Acrobleps hygrophilis* Hickman, 1979	17	0	4	0	21	0	17	4	21
	Mysmena sp. 1	1	0	0	0	1	0	1	0	1
	Mysmena sp. 2	3	0	1	0	4	0	3	1	4
	Mysmena sp. 3	33	6	2	7	35	13	39	9	48
	Trogloneta sp. 1	3	2	0	0	3	2	5	0	5
	Trogloneta sp. 2	10	1	1	0	11	1	11	1	12
Nicodamidae	*Litodamus* sp. 1	2	0	0	0	2	0	2	0	2

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Novodamus nodatus (Karsch, 1878)	6	1	8	0	14	1	7	8	15
Orsolobidae	*Cornifalx insignis* Hickman, 1979	0	0	0	1	0	1	0	1	1
	Tasmanoonops sp. 1	0	0	4	0	4	0	0	4	4
Pararchaeidae	**Flavarchaea hickmani* Rix, 2006	7	1	9	0	16	1	8	9	17
Pisauridae	*Dolomedes facetus* L. Koch, 1876	15	23	0	0	15	23	38	0	38
	Dolomedes sp. 1	40	98	6	6	46	14	138	12	150
Prodidomidae	*Molycria* sp. 1	0	0	2	0	2	0	0	2	2
Salticidae	*Arasia* sp. 1	0	1	0	0	0	1	1	0	1
	Astia hariola L. Koch, 1879	0	0	1	0	1	0	0	1	1
	Clynotis? sp. 1	0	0	0	5	0	5	0	5	5
	Ligonipes sp. 1	0	0	0	1	0	1	0	1	1
	Maratus sp. 1	14	16	3	2	17	18	30	5	35
	Maratus sp. 2	1	0	1	1	2	1	1	2	3
	Maratus sp. 3	4	0	0	0	4	0	4	0	4
	Maratus sp. 4	0	3	1	0	1	3	3	1	4
	Maratus sp. 5	6	0	0	0	6	0	6	0	6
	Neon sp. 1	2	0	0	0	2	0	2	0	2
	Opisthoncus magnidens L. Koch, 1880	0	22	0	21	0	43	22	21	43
	Opisthoncus necator L Koch, 1881	0	1	0	0	0	1	1	0	1
	Opisthoncus sp. 1	0	131	0	19	0	15	131	19	150
	Prostheclina sp. 1	0	25	1	0	1	25	25	1	26
	Prostheclina sp. 2	0	0	2	0	2	0	0	2	2
	Prostheclina sp. 3	2	39	4	5	6	44	41	9	50
	Salticidae sp. 1	0	1	2	0	2	1	1	2	3
	Salticidae sp. 2	4	3	0	0	4	3	7	0	7
	Salticidae sp. 3	1	6	0	2	1	8	7	2	9
	Salticidae sp. 4	1	0	1	0	2	0	1	1	2
	Salticidae sp. 5	0	2	0	0	0	2	2	0	2
	Salticidae sp. 6	0	2	0	1	0	3	2	1	3
	Salticidae sp. 7	0	1	0	0	0	1	1	0	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Salticidae sp. 8	0	1	0	0	0	1	1	0	1
	Salticidae sp. 9	0	1	0	12	0	13	1	12	13
	Salticidae sp. 10	0	0	0	1	0	1	0	1	1
	Salticidae sp. 11	0	92	0	9	0	11	92	9	101
	Salticidae sp. 12	0	4	0	0	0	4	4	0	4
	Salticidae sp. 13	0	0	0	1	0	1	0	1	1
	Salticidae sp. 14	0	8	0	0	0	8	8	0	8
	Salticidae sp. 15	0	5	0	2	0	7	5	2	7
	Salticidae sp. 16	0	0	1	0	1	0	0	1	1
	Simaetha sp. 1	0	1	0	0	0	1	1	0	1
Sparassidae	*Isopedella* sp. 1	0	0	1	0	1	0	0	1	1
	Neosparassus sp. 1	0	5	1	60	1	65	5	61	66
Stiphidiidae	Stiphidiidae sp. 1	0	0	2	0	2	0	0	2	2
Tetragnathidae	*Leucauge* sp. 1	0	0	0	1	0	1	0	1	1
	Metinae sp. 1	0	0	0	1	0	1	0	1	1
	Tetragnatha sp. 1	0	90	0	680	0	77	90	680	770
	Tetragnatha valida Keyserling, 1887	2	186	1	224	3	41	188	225	413
Theridiidae	*Achaearanea* sp. 1	1	0	0	7	1	7	1	7	8
	Achaearanea sp. 2	4	0	0	3	4	3	4	3	7
	Achaearanea sp. 3	1	0	1	2	2	2	1	3	4
	Achaearanea sp. 4	1	0	0	0	1	0	1	0	1
	Achaearanea sp. 5	0	0	3	7	3	7	0	10	10
	Achaearanea sp. 6	0	0	0	3	0	3	0	3	3
	Achaearanea sp. 7	0	1	0	0	0	1	1	0	1
	Achaearanea sp. 8	1	0	0	1	1	1	1	1	2
	Achaearanea tepidariorum (C.L. Koch, 1841)	0	1	0	4	0	5	1	4	5
	Conopistha sp. 1	0	1	0	0	0	1	1	0	1
	Cryptachaea veruculata? (Urquhart, 1886)	1	0	0	0	1	0	1	0	1
	Dipoena sp. 1	4	3	1	0	5	3	7	1	8
	Dipoena sp. 2	0	2	0	0	0	2	2	0	2
	Dipoena sp. 3	0	77	0	11	0	88	77	11	88

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Episinus sp. 1	1	9	0	21	1	3	10	21	31
	Euryopis elegans Keyserling, 1890	0	0	0	4	0	4	0	4	4
	Euryopis sp. 1	0	0	1	0	1	0	0	1	1
	Hadrotasrinae sp. 1	0	0	2	0	2	0	0	2	2
	Phoroncidia cygnea (Hickman, 1951)	0	0	0	1	0	1	0	1	1
	Phoroncidia sp. 1	1	2	1	3	2	5	3	4	7
	Phoroncidia sp. 2	0	0	1	0	1	0	0	1	1
	Phoroncidia trituberculata (Hickman, 1951)	0	0	0	1	0	1	0	1	1
	Steatoda sp. 1	15	3	0	0	15	3	18	0	18
	Steatoda sp. 2	0	0	0	1	0	1	0	1	1
	Steatoda sp. 3	0	1	0	0	0	1	1	0	1
	Steatoda sp. 4	0	0	0	5	0	5	0	5	5
	Steatoda sp. 5	1	0	1	0	2	0	1	1	2
	Theridiidae sp. 1	1	0	0	2	1	2	1	2	3
	Theridiidae sp. 2	0	1	1	16	1	17	1	17	18
	Theridiidae sp. 3	5	1	7	9	12	1	6	16	22
	Theridiidae sp. 4	0	0	0	1	0	1	0	1	1
	Theridiidae sp. 5	0	1	0	1	0	2	1	1	2
	Theridiidae sp. 6	0	0	0	4	0	4	0	4	4
	Theridiidae sp. 7	0	1	0	2	0	3	1	2	3
	Theridiidae sp. 8	0	1	0	0	0	1	1	0	1
	Theridiidae sp. 9	0	1	0	0	0	1	1	0	1
	Theridiidae sp. 10	0	1	0	2	0	3	1	2	3
	Theridiidae sp. 11	0	1	1	0	1	1	1	1	2
	Theridiidae sp. 12	0	1	0	16	0	17	1	16	17
	Theridiidae sp. 13	0	0	0	1	0	1	0	1	1
	Theridiidae sp. 14	0	0	0	1	0	1	0	1	1
	Theridiidae sp. 15	0	1	1	0	1	1	1	1	2
	Theridiidae sp. 16	1	0	0	0	1	0	1	0	1
	Theridiidae sp. 17	0	0	0	1	0	1	0	1	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Theridiidae sp. 18	0	0	0	1	0	1	0	1	1
	Theridiidae sp. 19	0	0	0	9	0	9	0	9	9
	Theridiidae sp. 20	0	0	9	0	9	0	0	9	9
	*Theridion limitatum*L. Koch, 1872	0	0	1	0	1	0	0	1	1
	Tobesoa sp. 1	0	3	0	7	0	1	3	7	10
Thomisidae	*Cymbacha ocellata* L. Koch, 1874	0	0	0	4	0	4	0	4	4
	Cymbacha sp. 1	1	5	0	3	1	8	6	3	9
	Diaea multopunctata L. Koch, 1874	0	2	0	4	0	6	2	4	6
	Diaea rosea L. Koch, 1875	1	860	0	191	1	151	861	191	1052
	Diaea sp. 1	0	42	0	7	0	49	42	7	49
	Diaea sp. 2	0	5	0	0	0	5	5	0	5
	Diaea sp. 3	0	0	0	2	0	2	0	2	2
	Diaea sp. 4	0	0	0	1	0	1	0	1	1
	Diaea sp. 5	0	0	0	1	0	1	0	1	1
	Diaea sp. 6	0	1	0	0	0	1	1	0	1
	Diaea sp. 7	0	1	0	0	0	1	1	0	1
	Diaea sp. 8	0	0	0	1	0	1	0	1	1
	Diaea sp. 9	0	2	0	23	0	25	2	23	25
	Diaea sp. 10	0	14	1	41	1	55	14	42	56
	Diaea sp. 11	0	2	0	8	0	1	2	8	10
	Sidymella sp. 1	0	0	1	0	1	0	0	1	1
	Sidymella sp. 2	3	0	1	6	4	6	3	7	10
	Sidymella sp. 3	0	1	0	2	0	3	1	2	3
	Sidymella sp. 4	0	5	1	0	1	5	5	1	6
	Sidymella sp. 5	1	0	0	0	1	0	1	0	1
	Sidymella sp. 6	0	9	1	9	1	18	9	10	19
	Stephanopis sp. 1	4	0	3	2	7	2	4	5	9
	Tharpyna sp. 1	0	2	0	4	0	6	2	4	6
Zodariidae	*Asteron reticulatum*? Jocqué, 1991	4	0	10	0	14	0	4	10	14
	**Habronestes driesseni* Baehr & Raven, 2009	12	0	2	0	14	0	12	2	14
	Zodariidae sp. 1	1	0	0	0	1	0	1	0	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Zodariidae sp. 3	2	0	0	0	2	0	2	0	2
	Zodariidae sp. 4	1	0	0	0	1	0	1	0	1
Zoridae	*Argoctenus pectinatus*? Hogg, 1900	1	1	16	0	17	1	2	16	18
	Argoctenus sp. 1	9	0	11	0	2	0	9	11	20
	Argoctenus sp. 2	0	0	1	0	1	0	0	1	1
	Elassoctenus sp. 1	0	0	1	0	1	0	0	1	1
	Hestimodema sp. 1	35	0	5	3	4	3	35	8	43
	Hestimodema sp. 2	12	0	8	3	2	3	12	11	23
	Thasyrhea sp. 1	0	3	0	2	0	5	3	2	5
Total Araneae		1049	3086	1645	3698	2252	3751	4135	5343	9478
Opilionida (Michael Driessen and Jonah Gouldethorpe)										
Monoscutidae	Monoscutidae sp. 1	85	0	34	0	119	0	85	34	119
Triaenonychidae	*Allonuncia grandis* Hickman, 1958	0	0	3	0	3	0	0	3	3
	Lomanella atrolutea Roewer, 1915	1	0	0	0	1	0	1	0	1
	Nunciella badia (Hickman, 1958)	7	0	26	0	33	0	7	26	33
	Nuncioides dysmicus Hickman, 1958	5	0	2	0	7	0	5	2	7
	Paranuncia gigantea Roewer, 1915	1	0	0	0	1	0	1	0	1
	Triaenonychidae sp. 1	0	0	1	0	1	0	0	1	1
	Triaenonychidae indet.	4	0	8	0	12	0	4	8	12
Total Opilionida		103	0	74	0	177	0	103	74	177
MALACOSTRACA										
Amphipoda (Alastair Richardson)										
Neoniphargidae	*Neoniphargus* sp. 1	2	0	1	0	3	0	2	1	3
Talitridae	*Austrotroides longicornis* Friend, 1987	1	0	0	0	1	0	1	0	1
	Keratroides vulgaris (Friend, 1979)	16	0	413	0	429	0	16	413	429
	Neorchestia plicibrancha Friend, 1987	115	0	100	0	215	0	115	100	215

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Orchestiella neambulans Friend, 1987	1	0	1	0	2	0	1	1	2
Total Amphipoda		135	0	515	0	650	0	135	515	650
Decapoda (Alastair Richardson)										
Parastacidae	*Ombrastacoides huonensis* Hansen & Richardson, 2006	15	0	0	0	15	0	15	0	15
	Ombrastacoides leptomerus (Riek, 1951)	1	0	4	0	5	0	1	4	5
	Ombrastacoides pulcher (Riek, 1967)	16	0	0	0	16	0	16	0	16
	Spinastacoides catinipalma Hansen & Richardson, 2006	1	0	0	0	1	0	1	0	1
	*Spinastacoides insignis (*Clark, 1939)	5	0	0	0	5	0	5	0	5
Total Decapoda		38	0	4	0	42	0	38	4	42
Isopoda (Allison Green)										
Philosciidae	*Plymophiloscia thomsoni* Green, 1961	0	0	2	0	2	0	0	2	2
Phreatoicidae	*Uramphisopus australis*	0	0	5	0	5	0	0	5	5
Styloniscidae	*Styloniscus maculosus* Green, 1961	4	0	2	0	6	0	4	2	6
	Styloniscus sp. 2	31	0	29	0	6	0	31	29	60
	Styloniscus squarrosus Green, 1961	8	0	16	0	24	0	8	16	24
Total Isopoda		43	0	54	0	43	0	43	54	97
INSECTA										
Blattodea (David Rentz)										
Blattellidae	*Johnrehnia*? sp. 1	8	0	4	0	12	0	8	4	12
Blattidae	*Platyzosteria* sp. 1	0	0	2	0	2	0	0	2	2
Total Blattodea		8	0	6	0	14	0	8	6	14

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
Coleoptera (Peter McQuillan)										
Anobiidae										
	Anobiidae sp. 1	0	4	0	7	0	11	4	7	11
	Anobiidae sp. 2	0	0	0	1	0	1	0	1	1
	Anobiidae sp. 3	0	1	0	0	0	1	1	0	1
Anthicidae	Anthicidae sp. 1	0	0	17	0	17	0	0	17	17
	Anthicus sp. 1	7	6	25	22	32	28	13	47	60
	Anthelephila denisonii (King, 1869)	6	0	0	0	6	0	6	0	6
Anthribidae	Anthribidae sp. 1	0	24	0	0	0	24	24	0	24
Apionidae	Apionidae sp. B	0	0	1	0	1	0	0	1	1
Belidae	*Belus bimaculatus* (Pascoe, 1871)	0	1	0	0	0	1	1	0	1
Cantharidae	*Chauliognathus lugubris* (Fabricius, 1801)	0	0	0	1	0	1	0	1	1
	Chauliognathus nobilitatus (Erichson, 1842)	0	4	0	1	0	5	4	1	5
	Chauliognathus spp. (L)	38	0	15	0	53	0	38	15	53
Carabidae	*Amblytelus* sp. 1	4	2	0	0	4	2	6	0	6
	Bembidion sp. 1	0	0	32	0	32	0	0	32	32
	Carabidae sp 3 (L)	0	0	1	0	1	0	0	1	1
	Carabidae sp. 1 (L)	0	0	3	0	3	0	0	3	3
	Carabidae sp. 2 (L)	1	0	0	0	1	0	1	0	1
	Harpalinae sp. 1	0	0	1	0	1	0	0	1	1
	Homethes sp. 1	1	0	1	0	2	0	1	1	2
	Lecanomerus sp. 1	0	0	26	2	26	2	0	28	28
	Promecoderus viridiaeneus Sloane, 1915	0	0	12	1	12	1	0	13	13
	Promecoderus viridiaeneus (L)	1	0	2	0	3	0	1	2	3
	Prosopogmus sp. 1	0	0	5	0	5	0	0	5	5
	Psydrini sp. 1	0	0	19	0	19	0	0	19	19
	Pterostichini sp. 1	0	0	2	0	2	0	0	2	2
	Scopodes spp.	8	0	55	0	63	0	8	55	63
	Trechini sp. 1	1	0	14	0	15	0	1	14	15
Cerambycidae	Cerambycidae sp. 1	0	2	0	0	0	2	2	0	2

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
Chrysomelidae	*Altica* sp. 1	1	11	1	17	2	28	12	18	30
	Altica sp. 2	0	25	2	11	2	36	25	13	38
	Altica sp. 3	0	0	0	4	0	4	0	4	4
	Altica sp. 4	0	2	1	1	1	3	2	2	4
	Altica sp. 5	3	0	2	2	5	2	3	4	7
	Altica sp. 6	0	0	1	0	1	0	0	1	1
	Altica sp. 7	2	0	8	0	1	0	2	8	10
	Calomela sp. 1	0	0	1	0	1	0	0	1	1
	Chrysomelidae sp. 1	0	10	1	0	1	1	10	1	11
	Chrysomelidae sp. 2	0	0	0	3	0	3	0	3	3
	Chrysomelidae sp. 4	0	0	2	0	2	0	0	2	2
	Cryptocephalinae sp. 1	0	8	0	0	0	8	8	0	8
	Cryptocephalinae sp. 2	0	0	0	1	0	1	0	1	1
	Cryptocephalus sp. 1	0	1	0	0	0	1	1	0	1
	Cryptocephalus sp. 2	0	2	0	0	0	2	2	0	2
	Edosa sp. 1	0	0	0	1	0	1	0	1	1
	Monolepta sp. 1	2	437	7	8	9	445	439	15	454
	Paropsionae (L)	0	5	0	0	0	5	5	0	5
	Paropsis sp. 1	0	1	0	0	0	1	1	0	1
	Paropsis sp. 2	0	3	0	1	0	4	3	1	4
	Paropsisterna bimaculata (Olivier, 1807)	0	1	0	8	0	9	1	8	9
	Paropsisterna lineata (Marsham, 1808)	0	5	0	0	0	5	5	0	5
	Paropsisterna sp. 1	1	4	0	0	1	4	5	0	5
	Paropsisterna sp. 2	0	2	0	0	0	2	2	0	2
	Chrysomelidae indet.	0	0	1	0	1	0	0	1	1
Cleridae	*Allelidea* sp. 1	0	2	0	1	0	3	2	1	3
	Cleridae sp. 1	0	3	0	0	0	3	3	0	3
	Cleridae sp. 2	0	0	0	1	0	1	0	1	1
Coccinellidae	*Rhyzobius* (L)	1	12	2	9	3	21	13	11	24
	Rhyzobius sp. 1	2	2	1	4	3	6	4	5	9
	Rhyzobius sp. 2	0	2	0	0	0	2	2	0	2

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Scymnus (L)	0	2	0	15	0	17	2	15	17
	Scymnus sp. 1	0	0	0	1	0	1	0	1	1
Corylophidae	Corylophidae sp. 1	0	1	0	0	0	1	1	0	1
Curculionidae	*Ancyttalia* sp. 1	0	21	0	2	0	23	21	2	23
	Ancyttalia sp. 2	0	1	0	0	0	1	1	0	1
	Aoplocnemis sp. 1	1	4	0	1	1	5	5	1	6
	Atelicus atrophus Pascoe, 1870	0	0	0	1	0	1	0	1	1
	Baris sp. 1	0	2	0	2	0	4	2	2	4
	Curculionidae sp. 3	0	0	0	3	0	3	0	3	3
	Curculionidae sp. 5	0	0	0	2	0	2	0	2	2
	Curculionidae sp. 6	0	1	0	0	0	1	1	0	1
	Curculionidae sp. 7	0	0	0	1	0	1	0	1	1
	Curculionidae sp. 8	0	0	0	2	0	2	0	2	2
	Curculionidae sp. 10	0	0	0	2	0	2	0	2	2
	Curculionidae sp. 14	2	0	0	1	2	1	2	1	3
	Curculionidae sp. 15	0	0	2	0	2	0	0	2	2
	Curculionidae sp. 16	1	0	0	0	1	0	1	0	1
	Haplonyx sp. 1	0	1	0	2	0	3	1	2	3
	Leptopius sp. 1	0	0	2	0	2	0	0	2	2
	Melanterius sp. 1	2	6	0	2	2	8	8	2	10
	Poropterus sp. 1	0	2	0	0	0	2	2	0	2
	Steriphus sp. 1	24	1	25	34	49	35	25	59	84
	Steriphus sp. 2	0	0	4	14	4	14	0	18	18
	Steriphus sp. 3	0	0	3	0	3	0	0	3	3
	Tychiini sp. 1	0	1	0	0	0	1	1	0	1
	Tychiini sp. 2	0	23	1	3	1	26	23	4	27
	Tychiini sp. 3	0	1	0	0	0	1	1	0	1
	Tychiini sp. 4	1	0	0	0	1	0	1	0	1
	Curculionidae indet. (L)	0	0	0	4	0	4	0	4	4
Elateridae	*Agrypnus victoriae* (Candèze, 1865)	0	0	2	0	2	0	0	2	2

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Conoderus sp. 1	1	0	0	1	1	1	1	1	2
	Crepidomerus sp. 1	0	1	0	0	0	1	1	0	1
	Elateridae sp. 1	0	0	0	1	0	1	0	1	1
	Elateridae sp. 2	0	0	0	10	0	1	0	10	10
	Elateridae sp. 3	0	0	0	2	0	2	0	2	2
	Parablax sp. 1	0	0	0	1	0	1	0	1	1
Elmidae	Elmidae indet. (L)	0	0	1	0	1	0	0	1	1
Erotylidae	Erotylidae sp. 1	0	0	0	1	0	1	0	1	1
Eucnemidae	Eucnemidae indent.	2	0	0	2	2	2	2	2	4
Histeridae	Histeridae sp. 1	0	1	0	0	0	1	1	0	1
Lathridiidae	*Corticaria* sp. 1	0	8	1	131	1	139	8	132	140
	Lathridiidae indet.	0	0	1	0	1	0	0	1	1
Leiodidae	Leiodidae sp. 1	1	0	1	0	2	0	1	1	2
	Nargomorphus sp. 1	2	0	12	1	14	1	2	13	15
	Pseudonemadus sp. 1	2	0	23	0	25	0	2	23	25
	Zeadolopus sp. 1	1	0	0	0	1	0	1	0	1
Lucanidae	*Lissotes* sp. nov.	2	0	0	0	2	0	2	0	2
Lycidae	*Porrostoma*? (L)	1	0	0	0	1	0	1	0	1
	Porrostoma sp. 1	0	3	3	32	3	35	3	35	38
	Porrostoma? sp. 2	0	0	0	0	0	0	0	0	0
	Trichalus sp. 1	1	4	0	12	1	16	5	12	17
Melandryidae	Melandryidae sp. A	0	0	0	5	0	5	0	5	5
	Melandryidae sp. B	0	1	0	0	0	1	1	0	1
	Melandryidae sp. C	0	0	0	1	0	1	0	1	1
	Orchesia sp. A	0	0	1	2	1	2	0	3	3
Melyridae	*Hypattalus* sp. 1	0	1	0	0	0	1	1	0	1
	Hypattalus sp. 1 (L)	0	0	1	0	1	0	0	1	1
	Melyridae sp. 1	0	0	0	5	0	5	0	5	5
	Melyridae sp. 2	0	1	0	41	0	42	1	41	42
Mordellidae	*Mordella* sp. 1	0	4	0	6	0	1	4	6	10
	Mordella sp. 2	0	1	0	0	0	1	1	0	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Mordella sp. 3	2	2	0	0	2	2	4	0	4
Nitidulidae	Nitidulidae sp. 1	0	1	0	0	0	1	1	0	1
	Thalycrodes sp. 1	10	0	17	2	27	2	10	19	29
Oedemeridae	*Ischnomera* sp. 1	0	1	0	0	0	1	1	0	1
	Sessinia lateralis (W.J. Macleay, 1887)	1	2	0	8	1	1	3	8	11
Phalacridae	Phalacridae sp. 1	0	3	0	5	0	8	3	5	8
Pselaphidae	*Euplectops* sp. 2	3	0	8	0	11	0	3	8	11
	Euplectops sp. 3	4	0	12	0	16	0	4	12	16
	Pselaphaulax sp. 1	16	0	2	0	18	0	16	2	18
	Pselaphidae sp. 1	0	0	2	0	2	0	0	2	2
	Pselaphidae sp. 2	0	0	1	0	1	0	0	1	1
	Pselaphidae sp. 3	1	0	1	0	2	0	1	1	2
	Pselaphidae sp. 5	0	0	4	0	4	0	0	4	4
	Pselaphidae sp. 6	3	0	0	0	3	0	3	0	3
	Pselaphidae sp. 8	1	0	18	0	19	0	1	18	19
	Pselaphidae sp. 9	5	0	3	0	8	0	5	3	8
	Pselaphidae sp. 10	0	0	2	0	2	0	0	2	2
	Pselaphidae sp. 11	0	0	1	0	1	0	0	1	1
	Pselaphidae sp. 12	0	0	1	0	1	0	0	1	1
	Rybaxis sp. 1	1	0	22	0	23	0	1	22	23
	Rybaxis sp. 2	0	0	4	0	4	0	0	4	4
	Rybaxis sp. 3	0	0	2	0	2	0	0	2	2
	Tasmanityrus sp. 1	2	0	0	0	2	0	2	0	2
	Tasmanityrus sp. 2	3	0	0	0	3	0	3	0	3
Psephenidae	*Sagola* sp. 1	3	0	4	0	7	0	3	4	7
	Sclerocyphon sp. 1	0	1	0	0	0	1	1	0	1
Ptilidae	Ptilidae sp. 1	0	0	1	0	1	0	0	1	1
Ptinidae	Ptinidae sp. 1	1	0	0	0	1	0	1	0	1
Salpingidae	*Notosalpingus* sp. 1	0	0	0	1	0	1	0	1	1
Scarabaeidae	*Heteronyx* sp. 1	0	0	0	1	0	1	0	1	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
Scirtidae	*Cyphon* sp. 1	0	35	0	38	0	73	35	38	73
	Macrohelodes crassus Blackburn, 1892	0	5	0	10	0	15	5	10	15
	Macrohelodes montanus Lea 1919	1	1	1	7	2	8	2	8	10
	Macrohelodes sp. 2	0	0	0	17	0	17	0	17	17
	Macrohelodes sp. 3	5	222	1	30	6	252	227	31	258
	Pseudomicrocara sp. 1	3	397	7	150	1	547	400	157	557
	Pseudomicrocara sp. 2	0	0	0	96	0	96	0	96	96
	Scirtidae sp. 1 (L)	0	0	1	0	1	0	0	1	1
	Scirtidae sp. 2 (L)	3	0	2	0	5	0	3	2	5
	Scirtidae sp. 3 (L)	0	0	4	0	4	0	0	4	4
	Scritidae sp. 4	3	64	0	0	3	64	67	0	67
	Scirtidae indet.	3	5	0	26	3	31	8	26	34
	Scirtidae indet. (L)	0	0	2	0	2	0	0	2	2
Scydmaenidae	*Euconnus* sp. 1	2	0	40	2	42	2	2	42	44
	Euconnus sp. 2	0	0	1	0	1	0	0	1	1
	Horaeomorphus sp. 1	0	0	56	0	56	0	0	56	56
Silvanidae	Silvanidae sp. 1	0	1	0	0	0	1	1	0	1
	Silvanidae sp. 2	0	1	0	0	0	1	1	0	1
	Silvanidae sp. 3	0	0	0	1	0	1	0	1	1
Staphylinidae	Aleochaerinae indet.	17	8	34	1	51	9	25	35	60
	Aleochaerinae sp. 1	11	0	20	0	31	0	11	20	31
	Aleochaerinae sp. 2	0	7	0	30	0	37	7	30	37
	Aleochaerinae sp. 3	0	0	1	0	1	0	0	1	1
	Aleochaerinae sp. 4	2	0	2	0	4	0	2	2	4
	Aleochaerinae sp. 5	1	0	0	0	1	0	1	0	1
	Anotylus sp. 1	1	1	2	0	3	1	2	2	4
	Euaesthetinae indet.	2	0	1	0	3	0	2	1	3
	Eulissus sp. 1	0	0	2	0	2	0	0	2	2
	Falagria sp. 1	3	0	1	0	4	0	3	1	4
	Omaliinae indet.	1	0	0	0	1	0	1	0	1
	Proteininae sp. 1	0	0	1	0	1	0	0	1	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Quedius sp. 1	1	0	96	0	97	0	1	96	97
	Quedius sp. 2	11	0	0	0	11	0	11	0	11
	Quedius sp. 3	0	0	1	0	1	0	0	1	1
	Scaphisoma sp. 1	3	0	0	0	3	0	3	0	3
	Tachinus sp. 1	4	0	0	0	4	0	4	0	4
	Staphylinidae indet.	0	0	4	0	4	0	0	4	4
	Staphylinoidea indet. (L)	1	0	11	0	12	0	1	11	12
Sylphidae	*Ptomaphila lacrymosa* (Schreibers, 1802)	1	0	0	0	1	0	1	0	1
Tenebrionidae	*Isopteron*? sp. 1	0	0	1	0	1	0	0	1	1
	Tenebrionidae indet. (L)	0	0	4	0	4	0	0	4	4
Cucujoidea indet. (L)	Cucujoidea indet. (L)	0	18	6	37	6	55	18	43	61
Coleoptera indet. (L)	Coleoptera indet. (L)	0	1	0	0	0	1	1	0	1
Total Coleoptera		258	1450	751	915	991	2329	1708	1666	3374
Dermaptera (Peter McQuillan)										
Anisolabididae	*Gonolabis tasmanicus* (Bormans, 1880)	0	0	3	0	3	0	0	3	3
Total Dermaptera		0	0	3	0	3	0	0	3	3
Diptera (Gunther Theischinger & Dan Bickel)										
Agromyzidae	Agromyzidae sp. 1	0	0	0	7	0	7	0	7	7
	Agromyzidae spp.	1	5	0	6	1	11	6	6	12
Anisopodidae	Anisopodidae sp. 1	30	0	199	1	229	1	30	200	230
	Anisopodidae spp.	7	3	3	0	1	3	10	3	13
	Anisopodidae spp. (L)	0	0	1	0	1	0	0	1	1
Anthomyzidae	Anthomyzidae spp.	3	21	4	0	7	21	24	4	28
Asilidae	Asilidae spp.	0	3	0	0	0	3	3	0	3
Bibionidae	Bibionidae sp. 1	0	0	0	9	0	9	0	9	9
Bombyliidae	Bombyliidae sp. 1	0	0	0	1	0	1	0	1	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Bombyliidae spp.	0	0	0	2	0	2	0	2	2
Calliphoridae	Calliphoridae sp. 1	0	0	0	2	0	2	0	2	2
	Calliphoridae sp. 2	0	1	0	0	0	1	1	0	1
	Calliphoridae sp. 3	1	0	0	24	1	24	1	24	25
	Calliphoridae sp. 4	0	0	0	1	0	1	0	1	1
	Calliphoridae spp.	7	15	1	64	8	79	22	65	87
Cecidomyiidae	Cecidomyiidae sp. 1	0	0	0	1	0	1	0	1	1
	Cecidomyiidae sp. 2	15	1	18	3	33	4	16	21	37
	Cecidomyiidae sp. 3	237	1	47	7	284	8	238	54	292
	Cecidomyiidae sp. 4	0	0	0	2	0	2	0	2	2
	Cecidomyiidae sp. 5	0	0	0	3	0	3	0	3	3
	Cecidomyiidae sp. 6	0	0	1	25	1	25	0	26	26
	Cecidomyiidae sp. 7	0	0	2	0	2	0	0	2	2
	Cecidomyiidae sp. 8	1	0	0	1	1	1	1	1	2
	Cecidomyiidae sp. 9	0	2	1	2	1	4	2	3	5
	Cecidomyiidae sp. 10	0	0	2	1	2	1	0	3	3
	Cecidomyiidae sp. 11	0	0	4	3	4	3	0	7	7
	Cecidomyiidae sp. 12	0	0	1	0	1	0	0	1	1
	Cecidomyiidae sp. 13	0	0	1	0	1	0	0	1	1
	Cecidomyiidae sp. 14	0	0	1	0	1	0	0	1	1
	Cecidomyiidae spp.	484	27	419	32	93	59	511	451	962
	Cecidomyiidae spp. (L)	4	0	6	0	1	0	4	6	10
Ceratopogonidae	Ceratopogonidae sp. 1	0	6	0	10	0	16	6	10	16
	Ceratopogonidae sp. 2	0	0	1	18	1	18	0	19	19
	Ceratopogonidae sp. 3	0	0	1	1	1	1	0	2	2
	Ceratopogonidae sp. 4	0	0	0	18	0	18	0	18	18
	Ceratopogonidae sp. 5	0	0	0	11	0	11	0	11	11
	Ceratopogonidae sp. 6	0	0	0	12	0	12	0	12	12
	Ceratopogonidae sp. 7	0	0	0	3	0	3	0	3	3
	Ceratopogonidae sp. 8	0	0	0	7	0	7	0	7	7
	Ceratopogonidae sp. 9	0	0	0	2	0	2	0	2	2

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Ceratopogonidae sp. 10	0	0	0	2	0	2	0	2	2
	Ceratopogonidae sp. 11	0	0	0	2	0	2	0	2	2
	Ceratopogonidae sp. 12	1	1	0	0	1	1	2	0	2
	Ceratopogonidae sp. 13	10	43	44	265	54	38	53	309	362
	Ceratopogonidae sp. 14	1	23	1	12	2	35	24	13	37
	Ceratopogonidae sp. 15	0	6	0	0	0	6	6	0	6
	Ceratopogonidae sp. 16	3	27	2	58	5	85	30	60	90
	Ceratopogonidae sp. 17	0	2	1	3	1	5	2	4	6
	Ceratopogonidae spp.	88	6767	78	1955	166	8722	6855	2033	8888
	Ceratopogonidae spp. (L)	0	0	5	0	5	0	0	5	5
Chamaemyiidae	Chamaemyiidae sp. 1	0	2	0	15	0	17	2	15	17
	Chamaemyiidae spp.	0	6	0	133	0	139	6	133	139
Chironomidae	Chironomidae sp. 1	0	0	0	1	0	1	0	1	1
	Chironomidae sp. 2	0	2	0	0	0	2	2	0	2
	Chironomidae sp. 3	0	2	2	0	2	2	2	2	4
	Chironomidae sp. 4	0	0	0	141	0	141	0	141	141
	Chironomidae sp. 5	0	0	1	0	1	0	0	1	1
	Chironomidae sp. 6	0	0	53	0	53	0	0	53	53
	Chironomidae sp. 7	0	0	4	0	4	0	0	4	4
	Chironomidae sp. 8	0	0	10	93	1	93	0	103	103
	Chironomidae sp. 9	0	0	0	180	0	18	0	180	180
	Chironomidae sp. 10	0	3	0	2	0	5	3	2	5
	Chironomidae sp. 11	0	0	0	2	0	2	0	2	2
	Chironomidae sp. 12	0	0	1	0	1	0	0	1	1
	Chironomidae sp. 13	0	1	0	0	0	1	1	0	1
	Chironomidae sp. 14	1	0	0	0	1	0	1	0	1
	Chironomidae sp. 15	2	77	15	163	17	24	79	178	257
	Chironomidae sp. 16	2	420	5	481	7	91	422	486	908
	Chironomidae sp. 17	0	15	2	145	2	16	15	147	162
	Chironomidae sp. 18	0	5	3	294	3	299	5	297	302

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Chironomidae sp. 19	0	4	0	1	0	5	4	1	5
	Chironomidae sp. 20	0	3	1	0	1	3	3	1	4
	Chironomidae sp. 21	0	14	1	6	1	2	14	7	21
	Chironomidae sp. 22	1	7	0	1	1	8	8	1	9
	Chironomidae sp. 23	4	286	12	96	16	382	290	108	398
	Chironomidae sp. 24	5	49	0	3	5	52	54	3	57
	Chironomidae sp. 25	0	7	0	24	0	31	7	24	31
	Chironomidae sp. 26	0	0	3	5	3	5	0	8	8
	Chironomidae sp. 27	0	0	17	14	17	14	0	31	31
	Chironomidae sp. 28	10	98	11	1169	21	1267	108	1180	1288
	Chironomidae sp. 29	0	0	0	1	0	1	0	1	1
	Chironomidae sp. 30	0	0	0	21	0	21	0	21	21
	Chironomidae sp. 31	0	1	2	22	2	23	1	24	25
	Chironomidae sp. 32	5	0	22	1	27	1	5	23	28
	Chironomidae sp. 33	2	3	0	0	2	3	5	0	5
	Chironomidae sp. 34	0	1	0	0	0	1	1	0	1
	Chironomidae sp. 35	0	1	0	0	0	1	1	0	1
	Chironomidae sp. 36	0	1	0	0	0	1	1	0	1
	Chironomidae sp. 37	0	1	0	0	0	1	1	0	1
	Chironomidae sp. 38	0	1	0	0	0	1	1	0	1
	Chironomidae spp.	90	610	93	1833	183	2443	700	1926	2626
	Chironomidae spp. (L)	1	0	7	0	8	0	1	7	8
Chloropidae	Chloropidae sp. 1	0	0	0	1	0	1	0	1	1
	Chloropidae sp. 2	0	0	0	3	0	3	0	3	3
	Chloropidae sp. 3	0	0	0	1	0	1	0	1	1
	Chloropidae sp. 4	0	0	0	14	0	14	0	14	14
	Chloropidae sp. 5	0	0	0	2	0	2	0	2	2
	Chloropidae sp. 6	0	0	0	6	0	6	0	6	6
	Chloropidae sp. 7	0	0	3	6	3	6	0	9	9
	Chloropidae sp. 8	0	0	0	1	0	1	0	1	1
	Chloropidae sp. 9	22	3	0	0	22	3	25	0	25

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Chloropidae sp. 10	3	0	0	0	3	0	3	0	3
	Chloropidae spp.	98	24	39	37	137	61	122	76	198
Conopidae	Conopidae sp. 1	0	0	0	1	0	1	0	1	1
Culicidae	Culicidae sp. 1	0	0	0	4	0	4	0	4	4
	Culicidae spp.	0	1	1	0	1	1	1	1	2
Dolichopodidae	*Chrysotimus* sp. 1	5	88	0	20	5	18	93	20	113
	Chrysotus parapicalis (Bickel & Dyte, 1989)	0	58	0	32	0	9	58	32	90
	Chrysotus sp. 1	0	2	0	1	0	3	2	1	3
	Dolichopodidae sp. 1	0	0	2	0	2	0	0	2	2
	Dolichopodidae sp. 2	0	0	1	0	1	0	0	1	1
	Dolichopodidae sp. 3	0	0	0	2	0	2	0	2	2
	Dolichopodidae sp. 4	0	0	1	0	1	0	0	1	1
	Dolichopodidae sp. 5	0	0	0	1	0	1	0	1	1
	Dolichopodidae sp. 6	1	0	0	0	1	0	1	0	1
	Dolichopodidae sp. 7	0	6	0	1	0	7	6	1	7
	Dolichopodidae sp. 8	0	1	0	0	0	1	1	0	1
	Dolichopodidae sp. 9	0	2	6	2	6	4	2	8	10
	Dolichopodidae sp. 10	0	0	7	0	7	0	0	7	7
	Dolichopodidae sp. 11	0	0	0	1	0	1	0	1	1
	Dolichopodidae sp. 12	0	1	0	0	0	1	1	0	1
	Kuringia sp. 1	13	2	3	0	16	2	15	3	18
	Sympycnus neboissi	0	7	0	1	0	8	7	1	8
	Sympycnus sp. 1	0	2	1	12	1	14	2	13	15
	Dolichopodidae spp.	87	179	25	16	112	195	266	41	307
Drosophilidae	Drosophilidae sp. 1	0	0	1	0	1	0	0	1	1
	Drosophilidae sp. 2	5	0	2	0	7	0	5	2	7
	Drosophilidae spp.	20	0	2	0	22	0	20	2	22
Empididae	*Apalocnemis* sp. 1	5	1	7	0	12	1	6	7	13
	**Hydropeza aptera* Sinclair 2016	14	0	0	0	14	0	14	0	14
	Cladodromia sp. 1	0	6	0	0	0	6	6	0	6

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Drapetis sp. 1	0	0	0	1	0	1	0	1	1
	Empididae sp. 1	1	0	1	0	2	0	1	1	2
	Empididae sp. 2	0	1	0	0	0	1	1	0	1
	Empididae sp. 3	0	0	0	2	0	2	0	2	2
	Empididae sp. 4	0	0	0	40	0	4	0	40	40
	Empididae sp. 5	0	0	0	3	0	3	0	3	3
	Empididae sp. 6	0	1	0	6	0	7	1	6	7
	Empididae sp. 7	0	0	0	1	0	1	0	1	1
	Empididae sp. 8	0	0	0	2	0	2	0	2	2
	Empididae sp. 9	0	0	0	1	0	1	0	1	1
	Empididae sp. 10	0	1	0	0	0	1	1	0	1
	Empididae sp. 11	0	2	0	0	0	2	2	0	2
	Empididae sp. 12	0	1	0	0	0	1	1	0	1
	Empididae sp. 13	0	2	0	2	0	4	2	2	4
	Empididae sp. 14	0	0	0	146	0	146	0	146	146
	Empididae sp. 15	0	0	0	1	0	1	0	1	1
	Empididae sp. 16	0	0	0	1	0	1	0	1	1
	Empididae sp. 17	0	0	0	1	0	1	0	1	1
	Empididae sp. 18	0	0	0	1	0	1	0	1	1
	Empis sp. 1	0	2	0	12	0	14	2	12	14
	Hilara sp. 1	0	14	0	0	0	14	14	0	14
	Hilaropus sp. 1	0	1	0	34	0	35	1	34	35
	Iteaphila? sp. 1	0	206	0	24	0	23	206	24	230
	Leptopeza sp. 1	1	0	0	0	1	0	1	0	1
	Tachydromia sp. 1	1	0	0	0	1	0	1	0	1
	Empididae spp.	8	556	29	644	37	12	564	673	1237
Ephydridae	Ephydridae sp. 1	0	3	17	12	17	15	3	29	32
	Ephydridae sp. 2	1	1	26	1	27	2	2	27	29
	Ephydridae sp. 3	0	0	0	7	0	7	0	7	7
	Ephydridae sp. 4	0	0	1	0	1	0	0	1	1
	Ephydridae sp. 5	0	0	1	0	1	0	0	1	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Ephydridae sp. 6	0	2	2	1	2	3	2	3	5
	Ephydridae spp.	1	4	4	0	5	4	5	4	9
Fergusoninidae	Fergusoninidae spp.	0	0	1	0	1	0	0	1	1
Heleomyzidae	Heleomyzidae sp. 1	0	1	0	0	0	1	1	0	1
	Heleomyzidae sp. 2	0	0	1	1	1	1	0	2	2
	Heleomyzidae sp. 3	0	1	11	8	11	9	1	19	20
	Heleomyzidae spp.	3	0	1	0	4	0	3	1	4
Helosciomyzidae	Helosciomyzidae sp. 1	0	0	1	3	1	3	0	4	4
	Helosciomyzidae sp. 2	0	0	10	102	1	12	0	112	112
	**Helosciomyza driesseni* McAlpine, 2012	0	0	0	8	0	8	0	8	8
Lauxaniidae	Lauxaniidae sp. 1	0	1	0	0	0	1	1	0	1
	Lauxaniidae sp. 2	0	0	0	4	0	4	0	4	4
	Lauxaniidae sp. 3	0	0	0	2	0	2	0	2	2
	Lauxaniidae sp. 4	0	0	0	2	0	2	0	2	2
	Lauxaniidae sp. 5	0	0	0	1	0	1	0	1	1
	Lauxaniidae sp. 6	0	0	0	2	0	2	0	2	2
	Lauxaniidae sp. 7	0	0	0	12	0	12	0	12	12
	Lauxaniidae spp.	2	12	1	37	3	49	14	38	52
Lonchaeidae	Lonchaeidae spp.	0	0	0	1	0	1	0	1	1
Lonchopteridae	Lonchopteridae sp. 1	0	0	0	10	0	1	0	10	10
Muscidae	Muscidae sp. 1	0	1	0	2	0	3	1	2	3
	Muscidae sp. 2	0	2	9	118	9	12	2	127	129
	Muscidae sp. 3	0	0	6	786	6	786	0	792	792
	Muscidae sp. 4	0	3	0	3	0	6	3	3	6
	Muscidae sp. 5	0	0	0	1	0	1	0	1	1
	Muscidae sp. 6	0	0	0	13	0	13	0	13	13
	Muscidae sp. 7	0	0	0	2	0	2	0	2	2
	Muscidae sp. 8	0	0	0	1	0	1	0	1	1
	Muscidae sp. 9	0	0	0	4	0	4	0	4	4
	Muscidae sp. 10	0	1	0	0	0	1	1	0	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Muscidae sp. 11	0	0	1	0	1	0	0	1	1
	Muscidae sp. 12	0	0	0	1	0	1	0	1	1
	Muscidae sp. 13	1	0	1	17	2	17	1	18	19
	Muscidae spp.	6	90	9	78	15	168	96	87	183
	Muscidae spp. (L)	121	0	11	0	132	0	121	11	132
Mycetophilidae	Mycetophilidae sp. 1	12	0	132	0	144	0	12	132	144
	Mycetophilidae sp. 2	7	4	0	0	7	4	11	0	11
	Mycetophilidae sp. 3	0	2	0	0	0	2	2	0	2
	Mycetophilidae sp. 4	0	1	0	0	0	1	1	0	1
	Mycetophilidae sp. 5	0	0	1	0	1	0	0	1	1
	Mycetophilidae sp. 6	5	0	0	0	5	0	5	0	5
	Mycetophilidae sp. 7	2	0	0	0	2	0	2	0	2
	Mycetophilidae sp. 8	0	0	2	0	2	0	0	2	2
	Mycetophilidae sp. 9	0	0	0	3	0	3	0	3	3
	Mycetophilidae sp. 10	0	0	0	1	0	1	0	1	1
	Mycetophilidae sp. 11	0	0	1	0	1	0	0	1	1
	Mycetophilidae sp. 12	0	0	1	0	1	0	0	1	1
	Mycetophilidae sp. 13	0	0	1	0	1	0	0	1	1
	Mycetophilidae sp. 14	0	0	1	0	1	0	0	1	1
	Mycetophilidae sp. 15	0	0	1	4	1	4	0	5	5
	Mycetophilidae sp. 16	0	0	0	2	0	2	0	2	2
	Mycetophilidae sp. 17	0	0	0	1	0	1	0	1	1
	Mycetophilidae sp. 18	0	0	0	1	0	1	0	1	1
	Mycetophilidae sp. 19	0	0	2	3	2	3	0	5	5
	Mycetophilidae sp. 20	1	0	0	0	1	0	1	0	1
	Mycetophilidae sp. 21	1	1	4	27	5	28	2	31	33
	Mycetophilidae sp. 22	1	0	0	0	1	0	1	0	1
	Mycetophilidae sp. 23	0	1	0	0	0	1	1	0	1
	Mycetophilidae sp. 24	0	1	0	0	0	1	1	0	1
	Mycetophilidae sp. 25	0	0	0	1	0	1	0	1	1
	Mycetophilidae sp. 26	0	0	0	2	0	2	0	2	2

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Mycetophilidae sp. 27	0	0	0	1	0	1	0	1	1
	Mycetophilidae spp.	8	13	165	28	173	41	21	193	214
Perissommatidae	Perissommatidae sp. 1	0	0	0	1	0	1	0	1	1
Phoridae	Phoridae sp. 1	4	0	16	5	2	5	4	21	25
	Phoridae sp. 2	4	0	0	0	4	0	4	0	4
	Phoridae sp. 3	18	2	22	61	4	63	20	83	103
	Phoridae sp. 4	0	0	0	1	0	1	0	1	1
	Phoridae sp. 5	136	5	150	78	286	83	141	228	369
	Phoridae sp. 6	1	0	1	0	2	0	1	1	2
	Phoridae sp. 7	1	0	0	0	1	0	1	0	1
	Phoridae sp. 8	3	0	7	5	1	5	3	12	15
	Phoridae sp. 9	2	0	0	0	2	0	2	0	2
	Phoridae sp. 10	0	0	1	0	1	0	0	1	1
	Phoridae spp.	270	60	393	72	663	132	330	465	795
Pipunculidae	Pipunculidae spp.	0	1	0	2	0	3	1	2	3
Platystomatidae	Platystomatidae spp.	0	0	0	1	0	1	0	1	1
Psychodidae	Psychodidae sp. 1	17	0	190	3	27	3	17	193	210
	Psychodidae sp. 2	21	0	4	0	25	0	21	4	25
	Psychodidae spp.	15	1	11	0	26	1	16	11	27
	Psychodidae spp. (L)	0	0	3	0	3	0	0	3	3
Rhagionidae	Rhagionidae sp. 1	0	0	0	4	0	4	0	4	4
	Rhagionidae spp.	0	328	0	251	0	579	328	251	579
Scatopsidae	Scatopsidae sp. 1	0	1	0	3	0	4	1	3	4
	Scatopsidae spp.	0	1	0	5	0	6	1	5	6
Sciaridae	Sciaridae sp. 1	20	8	59	29	79	37	28	88	116
	Sciaridae sp. 2	0	0	3	5	3	5	0	8	8
	Sciaridae sp. 3	6	2	1	0	7	2	8	1	9
	Sciaridae sp. 4	0	3	1	15	1	18	3	16	19
	Sciaridae sp. 5	13	2	1	1	14	3	15	2	17
	Sciaridae sp. 6	0	1	0	0	0	1	1	0	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Sciaridae sp. 7	1	0	0	0	1	0	1	0	1
	Sciaridae sp. 8	3	0	2	0	5	0	3	2	5
	Sciaridae spp.	33	50	132	88	165	138	83	220	303
Sciomyzidae	Sciomyzidae spp.	0	0	2	0	2	0	0	2	2
Simuliidae	Simuliidae sp. 1	0	56	0	27	0	83	56	27	83
	Simuliidae spp.	0	6	0	5	0	11	6	5	11
Sphaeroceridae	Sphaeroceridae sp. 1	11	0	363	4	374	4	11	367	378
	Sphaeroceridae sp. 2	1	0	12	1	13	1	1	13	14
	Sphaeroceridae sp. 3	2	0	20	1	22	1	2	21	23
	Sphaeroceridae sp. 4	10	0	0	0	1	0	10	0	10
	Sphaeroceridae sp. 5	11	0	15	0	26	0	11	15	26
	Sphaeroceridae sp. 6	2	0	5	0	7	0	2	5	7
	Sphaeroceridae sp. 7	0	0	0	1	0	1	0	1	1
	Sphaeroceridae sp. 8	0	0	1	0	1	0	0	1	1
	Sphaeroceridae spp.	144	0	229	0	373	0	144	229	373
Stratiomyidae	Stratiomyidae sp. 1	0	0	0	1	0	1	0	1	1
	Stratiomyidae spp.	0	1	1	1	1	2	1	2	3
	Stratiomyidae spp. (L)	1	0	0	0	1	0	1	0	1
Syrphidae	Syrphidae sp. 1	0	0	0	3	0	3	0	3	3
	Syrphidae sp. 2	0	0	0	20	0	2	0	20	20
	Syrphidae sp. 3	0	1	0	0	0	1	1	0	1
	Syrphidae sp. 4	0	3	0	0	0	3	3	0	3
	Syrphidae spp.	0	9	0	32	0	41	9	32	41
Tabanidae	Tabanidae sp. 1	0	5	1	13	1	18	5	14	19
	Tabanidae sp. 2	0	1	0	1	0	2	1	1	2
	Tabanidae sp. 3	0	0	0	1	0	1	0	1	1
	Tabanidae sp. 4	0	0	0	1	0	1	0	1	1
	Tabanidae sp. 5	0	0	0	1	0	1	0	1	1
	Tabanidae sp. 6	0	1	0	0	0	1	1	0	1
	Tabanidae spp.	0	22	0	39	0	61	22	39	61
	Tabanidae spp. (L)	1	0	1	0	2	0	1	1	2

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
Tachinidae	Tachinidae sp. 1	0	1	0	15	0	16	1	15	16
	Tachinidae sp. 2	0	1	0	5	0	6	1	5	6
	Tachinidae sp. 3	1	0	0	0	1	0	1	0	1
	Tachinidae sp. 4	0	0	0	4	0	4	0	4	4
	Tachinidae sp. 5	0	0	0	2	0	2	0	2	2
	Tachinidae sp. 6	0	0	0	1	0	1	0	1	1
	Tachinidae sp. 7	0	0	0	1	0	1	0	1	1
	Tachinidae sp. 8	0	0	0	1	0	1	0	1	1
	Tachinidae sp. 9	0	0	0	2	0	2	0	2	2
	Tachinidae sp. 10	0	0	0	1	0	1	0	1	1
	Tachinidae sp. 11	0	0	0	1	0	1	0	1	1
	Tachinidae sp. 12	0	0	0	1	0	1	0	1	1
	Tachinidae sp. 13	0	0	0	1	0	1	0	1	1
	Tachinidae sp. 14	0	0	0	1	0	1	0	1	1
	Tachinidae sp. 15	0	0	1	1	1	1	0	2	2
	Tachinidae sp. 16	0	1	0	0	0	1	1	0	1
	Tachinidae spp.	4	33	2	41	6	74	37	43	80
Tephritidae	Tephritidae sp. 1	0	0	0	2	0	2	0	2	2
	Tephritidae sp. 2	0	0	0	2	0	2	0	2	2
	Tephritidae sp. 3	0	0	0	1	0	1	0	1	1
	Tephritidae spp.	0	51	8	184	8	235	51	192	243
Therevidae	Therevidae spp.	0	0	0	1	0	1	0	1	1
Tipulidae	Cylindrotominae sp. 1	0	0	0	1	0	1	0	1	1
	Diemenomyia sp. 1	0	0	8	173	8	173	0	181	181
	Epiphragma meridionalis Alexander, 1928	0	0	1	0	1	0	0	1	1
	Gynoplistia kaoota Theischinger, 1993	0	2	0	0	0	2	2	0	2
	Gynoplistia leai (Alexander, 1922)	0	0	1	8	1	8	0	9	9
	Gynoplistia sp. 3	0	0	0	4	0	4	0	4	4
	Gynoplistia sp. 4	0	0	1	0	1	0	0	1	1
	Gynoplistia sp. 5	0	0	0	1	0	1	0	1	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Hexatominae sp. 1	0	0	1	0	1	0	0	1	1
	Dicranomyia cuneata (Skuse, 1890)	0	0	1	0	1	0	0	1	1
	Limnophila sp. 1	6	14	0	1	6	15	20	1	21
	Limnophila sp. 2	0	0	1	6	1	6	0	7	7
	Limnophila sp. 3	0	0	1	0	1	0	0	1	1
	Limnophila sp. 4	0	0	0	7	0	7	0	7	7
	Limonia sp. 1	0	1	0	0	0	1	1	0	1
	Limonia sp. 2	0	1	3	31	3	32	1	34	35
	Limoniinae sp. 1	2	0	0	0	2	0	2	0	2
	**Molophilus driesseni* Theischninger, 2012	0	0	3	10	3	1	0	13	13
	Molophilus flavonotatus Skuse, 1890	1	80	4	33	5	113	81	37	118
	*Molophilus horakae*Theischinger, 1994	0	1	0	0	0	1	1	0	1
	Molophilus sp. 1	1	0	1	0	2	0	1	1	2
	Molophilus sp. 6	0	0	0	2	0	2	0	2	2
	Molophilus sp. 8	0	0	1	0	1	0	0	1	1
	Paralimnophila sp. 1	0	0	1	0	1	0	0	1	1
	Paralimnophila sp. 3	1	0	1	0	2	0	1	1	2
	Paralimnophila sp. 4	0	0	0	3	0	3	0	3	3
	Paralimnophila sp. 5	0	0	0	1	0	1	0	1	1
	Paralimnophila sp. 6	0	0	2	3	2	3	0	5	5
	Tipulidae sp. 35	0	0	2	0	2	0	0	2	2
	Tipulinae sp. 1	0	11	0	3	0	14	11	3	14
	Tipulinae sp. 2	0	0	0	5	0	5	0	5	5
	Tipulinae sp. 3	0	0	0	1	0	1	0	1	1
	Tricyphona sp. 1	0	1	0	17	0	18	1	17	18
	Tipulidae spp.	44	120	75	224	119	344	164	299	463
	Tipulidae spp. (L)	2	0	22	0	24	0	2	22	24
Trichoceridae	Trichoceridae sp. 1	2	2	0	7	2	9	4	7	11
Diptera indet.	Diptera indet.	0	0	2	0	2	0	0	2	2
Total Diptera		2287	10779	3455	11415	4644	18738	13066	14870	27936

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
Hemiptera (Gerry Cassis & Celia Symonds)										
Auchenorrhyncha										
Achilidae	Achilidae sp. 38	0	0	0	1	0	1	0	1	1
Aphrophoridae	Aphrophoridae sp. 72	0	1	0	5	0	6	1	5	6
	Aphrophoridae sp. 96	0	0	0	0	0	0	0	0	0
	Aphrophoridae sp. 152	0	0	0	1	0	1	0	1	1
Cercopidae	Cercopidae sp. 22	0	0	0	185	0	185	0	185	185
Cicadellidae	Cicadellidae sp. 68	0	2	0	0	0	2	2	0	2
	Cicadellidae sp. 70	0	5	0	1	0	6	5	1	6
	Cicadellidae sp. 71	0	1	0	0	0	1	1	0	1
	Cicadellidae sp. 73	0	0	0	1	0	1	0	1	1
	Cicadellidae sp. 74	0	0	0	1	0	1	0	1	1
	Cicadellidae sp. 77	0	0	0	1	0	1	0	1	1
	Cicadellidae sp. 79	0	0	0	1	0	1	0	1	1
	Cicadellidae sp. 82	13	71	6	2	19	73	84	8	92
	Cicadellidae sp. 84	0	2	0	0	0	2	2	0	2
	Cicadellidae sp. 87	1	0	0	0	1	0	1	0	1
	Cicadellidae sp. 91	0	0	16	0	16	0	0	16	16
	Cicadellidae sp. 93	0	0	4	0	4	0	0	4	4
	Cicadellidae sp. 95	0	0	1	0	1	0	0	1	1
	Cicadellidae sp. 106	0	1	0	0	0	1	1	0	1
	Cicadellidae sp. 150	1	0	0	0	1	0	1	0	1
	Cicadellidae sp. 161	0	0	1	0	1	0	0	1	1
	Cicadellidae sp. 167	0	1	0	0	0	1	1	0	1
	Deltocephalinae sp. 9	0	9	0	0	0	9	9	0	9
	Deltacephalinae sp. 29	2	133	0	20	2	153	135	20	20
	Deltacephalinae sp. 40	0	0	9	1	9	1	0	10	10
	Deltacephalinae sp. 136	0	1	0	0	0	1	1	0	1
	Deltacephalinae sp. 142	1	18	2	5	3	23	19	7	26

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Deltacephalinae sp. 162	0	0	0	1	0	1	0	1	1
	Euacanthellinae sp. 20	2	0	3	1	5	1	2	4	6
	Euacanthellinae sp. 57	3	1	240	2	243	3	4	242	246
	Euacanthellinae sp. 66	6	1	3	0	9	1	7	3	10
	Idiocerinae sp. 26	0	5	0	0	0	5	5	0	5
	Idiocerinae sp. 44	0	0	0	0	0	0	0	0	0
	Idiocerinae sp. 51	0	1	0	0	0	1	1	0	1
	Lassinae? sp. 156	0	3	0	1	0	4	3	1	4
	Ledrinae sp. 43	0	4	0	0	0	4	4	0	4
	Ledrinae sp. 153	0	0	0	2	0	2	0	2	2
	Ledrinae sp. 157	0	0	0	1	0	1	0	1	1
	Ledrinae sp. 160	0	5	0	0	0	5	5	0	5
	Ledrinae sp. 164	0	1	0	0	0	1	1	0	1
	Macropsinae? sp. 166	0	1	0	0	0	1	1	0	1
	Tartessinae sp. 17	0	0	0	4	0	4	0	4	4
	Tartessinae sp. 23	0	0	0	0	0	0	0	0	0
	Tartessinae sp. 154	0	0	0	2	0	2	0	2	2
	Typhlocybinae sp. 42	1	80	0	2	1	82	81	2	83
	Typhlocybinae sp. 48	1	0	7	0	8	0	1	7	8
	Typhlocybinae sp. 62	0	0	0	1	0	1	0	1	1
	Typhlocybinae sp. 151	0	0	0	19	0	19	0	19	19
	Typhlocybinae sp. 158	0	0	2	3	2	3	0	5	5
	Ulopinae sp. 4	0	0	0	0	0	0	0	0	0
	Ulopinae sp. 5	6	237	5	848	11	185	243	853	1096
	Ulopinae sp. 18	3	1175	1	43	4	1218	1178	44	1222
	Ulopinae sp. 33	0	2	0	3	0	5	2	3	5
	Ulopinae sp. 34	0	1	1	3	1	4	1	4	5
	Ulopinae sp. 159	0	0	0	1	0	1	0	1	1
	Ulopinae sp. 168	0	0	1	0	1	0	0	1	1
	Xestocephalinae sp. 31	0	0	16	2	16	2	0	18	18
	Xestocephalinae sp. 32	1	1	4	1	5	2	2	5	7

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Xestocephalinae sp. 35	23	0	23	0	46	0	23	23	46
Cixiidae	Cixiidae sp. 88	0	1	0	0	0	1	1	0	1
	Cixiidae sp. 175	0	1	0	0	0	1	1	0	1
Delphacidae	Delphacidae sp. 16	19	0	10	18	29	18	19	28	47
	Delphacidae sp. 63	1	0	0	0	1	0	1	0	1
	Delphacidae sp. 177	2	0	0	1	2	1	2	1	3
	Delphacidae sp. 178	0	0	0	1	0	1	0	1	1
	Delphacidae sp. 179	1	0	0	1	1	1	1	1	2
	Delphacidae sp. 181	0	0	1	0	1	0	0	1	1
	Haplodelphax sp. 1	6	0	2	6	8	6	6	8	14
	Nilaparvata sp. nov.	1	0	0	0	1	0	1	0	1
Dictyopharidae	Dictyopharidae sp. 85	0	1	0	0	0	1	1	0	1
Flatidae	Flatidae sp. 163	0	1	0	0	0	1	1	0	1
Pachygronthidae	Pachygronthidae sp. 1	3	357	7	857	1	1214	360	864	1224
Total Auchenorrhyncha		97	2124	365	2049	453	3273	2221	2414	4500
Heteroptera										
Acanthosomatidae	Acanthosomatidae sp. 6	1	498	2	506	3	14	499	508	1007
	Acanthosomatidae sp. 53	0	3	0	0	0	3	3	0	3
	Acanthosomatidae sp. 104	0	332	0	45	0	377	332	45	377
	Acanthosomatidae sp. 107	0	20	0	1	0	21	20	1	21
	Acanthosomatidae sp. 113	0	1	0	0	0	1	1	0	1
Blissidae	Blissidae sp. 25	0	0	1	13	1	13	0	14	14
Ceratocombidae	Ceratocombidae sp. 52	79	1	37	0	116	1	80	37	117
Cymidae?	Cymidae? sp. 11	0	0	0	0	0	0	0	0	0
Enicocephalidae	*Systelloderis* sp. 61	35	0	14	0	49	0	35	14	49
Lygaeidae	Lygaeidae sp. 78	0	0	0	2	0	2	0	2	2
	Nysius sp. 12	1	2	0	36	1	38	3	36	39
Miridae	Miridae sp. 10	0	0	1	0	1	0	0	1	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Miridae sp. 55	0	3	0	0	0	3	3	0	3
	Miridae sp. 56	0	0	0	0	0	0	0	0	0
	Miridae sp. 69	0	6	0	39	0	45	6	39	45
	Miridae sp. 81	0	0	0	1	0	1	0	1	1
	Miridae sp. 90	4	1	0	0	4	1	5	0	5
	Miridae sp. 97	1	0	2	0	3	0	1	2	3
	Miridae sp. 110	0	0	0	1	0	1	0	1	1
	Miridae sp. 120	0	1	0	0	0	1	1	0	1
	Miridae sp. 121	0	2	0	0	0	2	2	0	2
Nabidae	Nabidae sp. 14	0	4	0	6	0	1	4	6	10
	Nabidae sp. 41	0	10	2	4	2	14	10	6	16
Ochteridae	Ochteridae sp. 36	1	0	22	0	23	0	1	22	23
Pachygronthidae	Pachygronthidae sp. 1	0	24	1	671	1	695	24	672	696
Pentatomidae	Pentatomidae sp. 7	0	12	2	158	2	17	12	160	172
	Pentatomidae sp. 21	0	1	0	0	0	1	1	0	1
	Pentatomidae sp. 46	0	1	0	0	0	1	1	0	1
	Pentatomidae sp. 58	0	1	0	0	0	1	1	0	1
	Pentatomidae sp. 111	5	0	0	0	5	0	5	0	5
	Pentatomidae sp. 112	0	1	0	0	0	1	1	0	1
Reduviidae	*Psuedobargylia* sp. 1	0	0	1	0	1	0	0	1	1
	Reduviidae sp. 37	0	0	2	0	2	0	0	2	2
	Reduviidae sp. 47	0	0	0	0	0	0	0	0	0
	Reduviidae sp. 76	1	0	1	3	2	3	1	4	5
	Reduviidae sp. 89	1	0	0	0	1	0	1	0	1
	Reduviidae sp. 103	0	0	0	0	0	0	0	0	0
Rhyparochromidae	*Laryngodus* sp. 87	0	1	0	0	0	1	1	0	1
	Rhyparochromidae sp. 115	0	0	0	1	0	1	0	1	1
	Rhyparochromidae sp. 116	0	1	0	1	0	2	1	1	2
	Rhyparochromidae sp. 117	0	0	2	1	2	1	0	3	3
	Rhyparochromidae sp. 118	0	6	0	0	0	6	6	0	6
	Rhyparochromidae sp. 119	1	0	0	0	1	0	1	0	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Rhyparochromidae sp. 122	1	0	0	0	1	0	1	0	1
	Tasmanicola sp. 8	0	0	1	2	1	2	0	3	3
	Tomocoris sp. 39	0	0	0	0	0	0	0	0	0
	Udeocoris sp. 60	0	0	1	0	1	0	0	1	1
Schizopteridae	Schizopteridae sp. 19	5	0	11	0	16	0	5	11	16
Tingidae	**Lasiacantha ginger* Cassis & Symonds, 2011	0	0	3	0	3	0	0	3	3
	Tingidae sp. 15	0	0	0	0	0	0	0	0	0
	Tingidae sp. 59	0	1	0	0	0	1	1	0	1
	Tingidae sp. 108	0	0	0	1	0	1	0	1	1
	Tingidae sp. 114	1	0	0	0	1	0	1	0	1
Veliidae	Veliidae sp. 99	1	0	1	0	2	0	1	1	2
Total Heteroptera		138	933	107	1492	245	1273	1071	1599	2670
Sternorrhyncha										
Psyllidae	Psyllidae sp. 24	0	0	0	4	0	4	0	4	4
	Psyllidae sp. 30	0	46	0	0	0	46	46	0	46
	Psyllidae sp. 45	0	8	1	7	1	15	8	8	16
	Psyllidae sp. 49	0	1	1	11	1	12	1	12	13
	Psyllidae sp. 50	0	15	1	70	1	85	15	71	86
	Psyllidae sp. 54	0	1	1	3	1	4	1	4	5
	Psyllidae sp. 67	0	0	0	5	0	5	0	5	5
	Psyllidae sp. 75	0	0	0	10	0	1	0	10	10
	Psyllidae sp. 126	4	157	2	13	6	17	161	15	176
	Psyllidae sp. 127	0	0	0	9	0	9	0	9	9
	Psyllidae sp. 128	0	13	1	3	1	16	13	4	17
	Psyllidae sp. 129	0	2	0	33	0	35	2	33	35
	Psyllidae sp. 131	0	0	1	10	1	1	0	11	11
	Psyllidae sp. 132	0	0	0	1	0	1	0	1	1
	Psyllidae sp. 134	0	0	0	1	0	1	0	1	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Psyllidae sp. 135	0	0	0	4	0	4	0	4	4
	Psyllidae sp. 137	0	0	0	2	0	2	0	2	2
	Psyllidae sp. 138	0	3	0	1	0	4	3	1	4
	Psyllidae sp. 139	0	0	0	1	0	1	0	1	1
	Psyllidae sp. 140	0	0	0	2	0	2	0	2	2
	Psyllidae sp. 141	0	0	0	1	0	1	0	1	1
	Psyllidae sp. 143	0	0	0	1	0	1	0	1	1
	Psyllidae sp. 144	0	0	0	1	0	1	0	1	1
	Psyllidae sp. 145	0	0	0	1	0	1	0	1	1
	Psyllidae sp. 147	0	1	0	4	0	5	1	4	5
	Psyllidae sp. 148	0	0	0	1	0	1	0	1	1
	Psyllidae sp. 149	0	0	0	3	0	3	0	3	3
	Psyllidae sp. 170	0	0	0	0	0	0	0	0	0
	Psyllidae sp. 171	0	0	0	2	0	2	0	2	2
	Psyllidae sp. 172	0	0	0	1	0	1	0	1	1
	Psyllidae sp. 173	0	0	0	1	0	1	0	1	1
	Psyllidae sp. 174	0	0	0	1	0	1	0	1	1
Triozidae	Triozidae sp. 124	0	1	0	0	0	1	1	0	1
	Triozidae sp. 125	0	1	0	0	0	1	1	0	1
	Triozidae sp. 130	0	0	1	73	1	73	0	74	74
	Triozidae sp. 133	0	0	0	1	0	1	0	1	1
	Triozidae sp. 146	0	0	0	1	0	1	0	1	1
	Triozidae sp. 64	0	0	0	3	0	3	0	3	3
Sternorrhyncha indet.	Sternorrhyncha indet.	34	286	117	345	151	631	320	462	782
Total Sternorrhyncha		38	535	126	630	164	994	573	756	1329
Hemiptera indet.	Hemiptera indet. nymph	38	507	170	199	28	76	545	369	914
Total Hemiptera		311	4099	768	4370	890	5616	4410	5138	9413

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
Hymenoptera (non-ants: Chris Burwell, Abbey Throssell & Kevin Bonham; ants: Steve Shattuck & Nicky Meeson)										
Anthophoridae	Anthophoridae spp.	0	1	0	0	0	1	1	0	1
Aphelinidae	Aphelinidae sp. 1	0	2	0	0	0	2	2	0	2
	Aphelinidae sp. 2	0	1	0	0	0	1	1	0	1
	Aphelinidae sp. 3	0	1	0	0	0	1	1	0	1
	Aphelinidae sp. 4	0	0	0	1	0	1	0	1	1
	Azotus sp. 1	0	1	0	0	0	1	1	0	1
	Neocasca sp. 1	1	0	0	0	1	0	1	0	1
	Neocasca sp. 2	0	3	0	0	0	3	3	0	3
	Aphelinidae spp.	6	40	3	32	9	72	46	35	81
Apidae	*Apis mellifera* Linnaeus, 1758	1	1	0	0	1	1	2	0	2
	Apidae spp.	2	16	1	11	3	27	18	12	30
Austroniidae	*Austronia nitida* Riek, 1955	0	0	0	1	0	1	0	1	1
Bethylidae	*Sierola* sp. 1	0	2	0	0	0	2	2	0	2
	Bethylidae spp.	0	7	0	2	0	9	7	2	9
Braconidae	"Rogadinae" sp. 1	0	1	0	2	0	3	1	2	3
	"Rogadinae" sp. 2	0	1	0	1	0	2	1	1	2
	"Rogadinae" sp. 3	0	10	0	32	0	42	10	32	42
	Bracon? sp. 1	0	1	0	0	0	1	1	0	1
	Agathidiinae sp. 1	0	1	0	0	0	1	1	0	1
	Asobara sp. 1	0	2	0	0	0	2	2	0	2
	Aspilota andyausitni? Wharton, 2002	0	0	0	1	0	1	0	1	1
	Braconinae sp. 1	0	1	0	7	0	8	1	7	8
	Braconinae sp. 2	0	1	0	0	0	1	1	0	1
	Dinotrema sp. 1	1	0	1	0	2	0	1	1	2
	Dolichogenidia sp. 1	0	1	0	6	0	7	1	6	7
	Eiopius sp. 1	0	0	0	3	0	3	0	3	3
	Hormiinae sp. 1	0	1	0	0	0	1	1	0	1
	Meteorus sp. 1	0	0	0	1	0	1	0	1	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Microctonus sp. 1	0	0	0	3	0	3	0	3	3
	Microgastrinae sp. 1	0	0	1	1	1	1	0	2	2
	Microgastrinae sp. 2	0	0	0	1	0	1	0	1	1
	Microgastrinae sp. 3	0	1	0	0	0	1	1	0	1
	Microgastrinae sp. 4	0	1	0	0	0	1	1	0	1
	Microgastrinae sp. 5	0	1	0	0	0	1	1	0	1
	Microgastrinae sp. 6	0	2	0	0	0	2	2	0	2
	Microgastrinae sp. 7	0	1	0	0	0	1	1	0	1
	Opiinae sp. 1	0	0	0	1	0	1	0	1	1
	Opius sp. 1	0	1	0	1	0	2	1	1	2
	Triaspis sp. 1	0	0	0	1	0	1	0	1	1
	Braconidae spp.	9	103	8	88	17	191	112	96	208
Ceraphronidae	*Aphonogmus* sp. 1	0	3	0	3	0	6	3	3	6
	Aphonogmus sp. 2	9	0	0	0	9	0	9	0	9
	Aphonogmus sp. 3	3	0	0	0	3	0	3	0	3
	Aphonogmus sp. 4	0	0	0	3	0	3	0	3	3
	Aphonogmus sp. 5	0	0	0	1	0	1	0	1	1
	Aphonogmus sp. 6	0	0	20	0	2	0	0	20	20
	Ceraphron sp. 1	0	0	1	0	1	0	0	1	1
	Ceraphron sp. 2	0	0	1	0	1	0	0	1	1
	Ceraphronidae spp.	11	0	9	0	2	0	11	9	20
Ceraphronoidea	Ceraphronoidea spp.	0	23	0	6	0	29	23	6	29
Chalcididae	Chalcididae spp.	0	1	0	1	0	2	1	1	2
Charipidae	Charipidae sp. 1	0	1	0	0	0	1	1	0	1
Colletidae	Colletidae sp. 1	0	1	0	0	0	1	1	0	1
Diapriidae	*Acanthobetyla*? sp.nov.	0	0	0	3	0	3	0	3	3
	Belytinae sp. 1	0	0	2	1	2	1	0	3	3
	Belytinae sp. 2	0	0	0	4	0	4	0	4	4
	Belytinae sp. 3	0	0	0	1	0	1	0	1	1
	Belytinae sp. 4	0	0	0	1	0	1	0	1	1
	Belytinae sp. 5	1	0	0	0	1	0	1	0	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Diphoropria compsodes Naumann, 1982	0	0	0	1	0	1	0	1	1
	Diphoropria sp. 1	0	0	1	0	1	0	0	1	1
	Entomacis sp. 1	0	0	2	1	2	1	0	3	3
	Entomacis sp. 2	0	0	3	1	3	1	0	4	4
	Entomacis sp. 3	1	0	0	5	1	5	1	5	6
	Idiotypa sp. 1	0	0	2	1	2	1	0	3	3
	nr *Idiotypa* sp. 1	0	0	8	1	8	1	0	9	9
	Paramesius sp. 1	0	0	1	2	1	2	0	3	3
	Paramesius sp. 2	1	1	0	0	1	1	2	0	2
	Pentapria sp. 1	1	0	0	0	1	0	1	0	1
	Perisodryas daedalma Turner, 1935	0	0	0	2	0	2	0	2	2
	Spilomicrus sp. 1	0	1	2	10	2	11	1	12	13
	Spilomicrus sp. 2	0	0	9	25	9	25	0	34	34
	Spilomicrus sp. 3	0	0	1	4	1	4	0	5	5
	Spilomicrus sp. 4	0	0	0	1	0	1	0	1	1
	Spilomicrus sp. 5	0	0	0	1	0	1	0	1	1
	Spilomicrus sp. 6	0	0	0	2	0	2	0	2	2
	Spilomicrus sp. 7	0	0	0	1	0	1	0	1	1
	Stylaclista sp. 1	0	0	1	1	1	1	0	2	2
	Stylaclista sp. 2	0	0	1	0	1	0	0	1	1
	Trichoporia sp. 1	1	0	0	0	1	0	1	0	1
	Trichoporia sp. 2	0	0	2	0	2	0	0	2	2
	Trichoporia sp. 3	0	0	0	1	0	1	0	1	1
	Trichoporia sp. 4	0	0	0	1	0	1	0	1	1
	Diapriidae spp.	35	28	111	42	146	7	63	153	216
Dryinidae	Dryinidae sp. 1	0	1	0	0	0	1	1	0	1
	Dryinidae spp.	0	4	0	14	0	18	4	14	18
Encyrtidae	Encyrtidae sp. 1	0	0	0	1	0	1	0	1	1
	Encyrtidae sp. 2	1	3	1	0	2	3	4	1	5
	Encyrtidae sp. 3	0	3	0	2	0	5	3	2	5

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Encyrtidae sp. 4	0	1	1	0	1	1	1	1	2
	Encyrtidae sp. 5	1	0	0	1	1	1	1	1	2
	Encyrtidae sp. 6	0	1	0	0	0	1	1	0	1
	Encyrtidae sp. 7	0	1	0	0	0	1	1	0	1
	Encyrtidae sp. 8	0	2	0	0	0	2	2	0	2
	Encyrtidae sp. 9	1	0	0	0	1	0	1	0	1
	Encyrtidae sp. 10	1	0	0	0	1	0	1	0	1
	Encyrtidae sp. 11	0	1	0	0	0	1	1	0	1
	Encyrtidae sp. 12	0	2	0	0	0	2	2	0	2
	Encyrtidae sp. 13	0	1	0	0	0	1	1	0	1
	Encyrtidae sp. 15	1	2	0	0	1	2	3	0	3
	Encyrtidae sp. 16	0	2	0	0	0	2	2	0	2
	Encyrtidae sp. 17	0	0	4	0	4	0	0	4	4
	Encyrtidae sp. 18	0	0	1	5	1	5	0	6	6
	Encyrtidae sp. 19	0	0	0	1	0	1	0	1	1
	Encyrtidae sp. 20	0	0	0	3	0	3	0	3	3
	Encyrtidae sp. 22	0	1	0	1	0	2	1	1	2
	Encyrtidae sp. 23	0	2	0	0	0	2	2	0	2
	Encyrtidae sp. 24	0	1	0	0	0	1	1	0	1
	Encyrtidae sp. 25	0	1	0	0	0	1	1	0	1
	Encyrtidae sp. 26	0	1	0	0	0	1	1	0	1
	Encyrtidae sp. 27	0	1	0	0	0	1	1	0	1
	Encyrtidae sp. 28	0	1	0	0	0	1	1	0	1
	Encyrtidae sp. 29	2	4	0	0	2	4	6	0	6
	Encyrtidae sp. 30	1	0	0	0	1	0	1	0	1
	Encyrtidae sp. 31	1	1	0	0	1	1	2	0	2
	Encyrtidae sp. 32	0	4	0	0	0	4	4	0	4
	Encyrtidae sp. 33	0	0	0	2	0	2	0	2	2
	Encyrtidae sp. 34	0	0	0	1	0	1	0	1	1
	Encyrtidae sp. 35	0	0	0	1	0	1	0	1	1
	Encyrtidae spp.	32	137	29	77	61	214	169	106	275

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
Eucoilidae	Eucoilidae sp. 1	0	0	4	0	4	0	0	4	4
	Eucoilidae sp. 2	0	0	0	1	0	1	0	1	1
	Eucoilidae sp. 3	0	0	0	1	0	1	0	1	1
	Eucoilidae sp. 4	0	0	0	1	0	1	0	1	1
Eulophidae	*Cirrospilus* sp. 1	0	0	0	1	0	1	0	1	1
	Cirrospilus sp. 2	0	0	0	1	0	1	0	1	1
	Diglyphus sp. 1	0	0	0	1	0	1	0	1	1
	Entedoninae sp. 1	0	0	0	1	0	1	0	1	1
	Entedoninae sp. 2	0	0	0	2	0	2	0	2	2
	Entedoninae sp. 3	0	0	0	1	0	1	0	1	1
	Entedoninae sp. 4	0	0	0	1	0	1	0	1	1
	Entedoninae sp. 5	0	0	0	2	0	2	0	2	2
	Entedoninae sp. 6	0	0	0	10	0	1	0	10	10
	Entedoninae sp. 7	0	0	0	1	0	1	0	1	1
	Entedoninae sp. 8	1	2	0	0	1	2	3	0	3
	Entedoninae sp. 9	0	1	0	0	0	1	1	0	1
	Entedoninae sp. 10	0	1	0	0	0	1	1	0	1
	Euderus sp. 1	1	2	0	3	1	5	3	3	6
	Eulophinae sp. 1	0	0	0	1	0	1	0	1	1
	Euplectrus sp. 1	0	2	0	0	0	2	2	0	2
	Hemiptarsenus varicornis (Girault, 1913)	0	0	0	2	0	2	0	2	2
	Naumanniola varians Boucek, 1988	0	2	1	2	1	4	2	3	5
	Ophelimus sp. 1	0	0	0	1	0	1	0	1	1
	Ophelimus sp. 2	0	0	0	1	0	1	0	1	1
	Ophelimus sp. 3	0	0	0	1	0	1	0	1	1
	Tetrastichinae sp. 1	0	3	0	1	0	4	3	1	4
	Tetrastichinae sp. 2	0	2	0	2	0	4	2	2	4
	Tetrastichinae sp. 3	1	3	0	1	1	4	4	1	5
	Tetrastichinae sp. 4	0	3	1	1	1	4	3	2	5
	Tetrastichinae sp. 6	0	0	0	1	0	1	0	1	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Tetrastichinae sp. 7	0	2	0	3	0	5	2	3	5
	Tetrastichinae sp. 8	0	0	0	1	0	1	0	1	1
	Eulophidae spp.	3	307	3	103	6	41	310	106	416
Eupelmidae	*Eupelmus* sp. 1	0	2	0	0	0	2	2	0	2
	Eupelmus sp. 2	0	1	0	0	0	1	1	0	1
Eurytomidae	*Eurytoma* sp. 1	0	0	0	1	0	1	0	1	1
	Eurytoma sp. 2	0	1	0	0	0	1	1	0	1
	Eurytomidae spp.	0	9	0	0	0	9	9	0	9
Evaniidae	Evaniidae sp. 1	1	0	0	0	1	0	1	0	1
Figitidae	Figitidae spp.	13	0	54	2	67	2	13	56	69
Formicidae	*Amblyopone australis* Erichson, 1842	2	0	0	0	2	0	2	0	2
	Anonychomyrma nitidiceps? (E. André, 1896)	1400	78	66	1	1466	79	1478	67	1545
	Austromorium flavigaster (Clark, 1938)	3	0	14	0	17	0	3	14	17
	Colobostruma elliotti	0	0	1	0	1	0	0	1	1
	Colobostruma sp. 1	2	0	0	0	2	0	2	0	2
	Hypoponera sp. 1	24	0	76	1	1	1	24	77	101
	Iridomyrmex sp. 1 (*anceps* group)	1379	7	639	1	218	8	1386	640	2026
	Monomorium sculpturatum Clark, 1934	13	0	22	0	35	0	13	22	35
	Monomorium sp. 1 (*leae* group.)	13	0	29	0	42	0	13	29	42
	Monomorium sp. 2	4	0	0	0	4	0	4	0	4
	Myrmecia sp. (*pilosula* group)	1	0	2	0	3	0	1	2	3
	Notoncus ectatommoides (Forel, 1892)	6	0	0	0	6	0	6	0	6
	Ochetellus sp. 1 (*glaber* group)	23	23	0	0	23	23	46	0	46
	Polyrhachis patiens Santschi, 1920	3	0	0	0	3	0	3	0	3
	Prolasius nitidissimus (E. André, 1896)	4	19	0	0	4	19	23	0	23
	Prolasius sp. 1 (*pallidus* group)	20	1	14	0	34	1	21	14	35
	Strumigenys perplexa (Smith, 1876)	12	0	0	0	12	0	12	0	12
	Technomyrmex albipes? (Smith, 1861)	0	0	2	0	2	0	0	2	2
Halictidae	*Lasioglossum* sp. 1	0	9	0	0	0	9	9	0	9
	Lasioglossum sp. 2	0	1	0	0	0	1	1	0	1
	Lasioglossum sp. 3	1	0	0	0	1	0	1	0	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Halictidae spp.	0	3	0	0	0	3	3	0	3
Ichneumonidae	Banchinae sp. 2	0	0	0	1	0	1	0	1	1
	Casinaria? sp. 1	0	0	0	1	0	1	0	1	1
	Debophanes areolatus? Gauld, 1984	0	0	2	0	2	0	0	2	2
	Isodromas sp. 1	0	0	0	1	0	1	0	1	1
	nr *Lissonota* sp. 1	0	0	0	1	0	1	0	1	1
	Orthocentrinae sp. 1	0	1	0	0	0	1	1	0	1
	Orthocentrinae sp. 2	0	0	0	1	0	1	0	1	1
	Paraphylax sp. 1	0	1	0	0	0	1	1	0	1
	Phygadeuontinae sp. 1	0	1	0	0	0	1	1	0	1
	Phygadeuontinae sp. 2	0	1	0	0	0	1	1	0	1
	Phygadeuontinae sp. 3	0	1	0	0	0	1	1	0	1
	Phygadeuontinae sp. 4	0	1	0	0	0	1	1	0	1
	Phygadeuontinae sp. 5	0	0	0	3	0	3	0	3	3
	Phygadeuontinae sp. 6	0	0	0	1	0	1	0	1	1
	Pimplinae sp. 1	0	0	0	1	0	1	0	1	1
	Venturia ocypeta Gauld 1984	0	2	0	1	0	3	2	1	3
	Trichoma biroi? Szépligeti, 1906	0	0	0	1	0	1	0	1	1
	Ichneumonidae spp.	4	116	3	38	7	154	120	41	161
Megaspilidae	Megaspilidae spp.	0	0	3	0	3	0	0	3	3
Mutillidae	Mutillidae sp. 1	2	0	0	0	2	0	2	0	2
	Mutillidae spp.	0	1	0	0	0	1	1	0	1
Mymaridae	Mymaridae sp. 1	1	0	10	0	11	0	1	10	11
	Mymaridae sp. 2	0	16	0	0	0	16	16	0	16
	Mymaridae sp. 3	1	0	0	0	1	0	1	0	1
	Mymaridae sp. 4	1	0	0	0	1	0	1	0	1
	Mymaridae sp. 5	0	0	1	0	1	0	0	1	1
	Mymaridae sp. 6	0	0	0	1	0	1	0	1	1
	Mymaridae sp. 7	0	0	0	1	0	1	0	1	1
	Mymaridae sp. 8	0	0	0	2	0	2	0	2	2

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Mymaridae sp. 9	0	0	0	1	0	1	0	1	1
	Mymaridae spp.	10	28	42	18	52	46	38	60	98
Pergidae	Pergidae spp.	0	0	0	1	0	1	0	1	1
Platygastridae	*Alfredella* sp. 1	0	1	0	0	0	1	1	0	1
	Allotropa sp. 1	0	0	0	1	0	1	0	1	1
	Allotropa sp. 2	0	1	0	0	0	1	1	0	1
	Allotropa sp. 3	0	1	0	0	0	1	1	0	1
	Fidiobia sp. 1	0	0	0	2	0	2	0	2	2
	Isostasius sp. 1	0	0	0	1	0	1	0	1	1
	Isostasius sp. 2	0	0	0	8	0	8	0	8	8
	Synopeas sp. 1	0	0	1	1	1	1	0	2	2
	Synopeas sp. 2	0	0	0	1	0	1	0	1	1
	Synopeas sp. 3	0	0	0	5	0	5	0	5	5
	Platygastridae spp.	0	36	0	18	0	54	36	18	54
Pompilidae	*Sphinctostethus* sp. 1	0	0	1	0	1	0	0	1	1
	Pompilidae spp.	2	1	0	2	2	3	3	2	5
Proctotrupidae	Proctotrupidae spp.	0	1	0	0	0	1	1	0	1
Pteromalidae	*Acroclisoides* sp. 1	1	0	0	1	1	1	1	1	2
	Agamerion sp. 1	0	0	1	0	1	0	0	1	1
	Amerostenus sp. 1	0	1	0	0	0	1	1	0	1
	Coelocyba sp. 1	0	0	0	1	0	1	0	1	1
	Dipara sp. 1	0	0	1	0	1	0	0	1	1
	Gastrancistrus sp. 1	0	0	1	4	1	4	0	5	5
	Ophelosia sp. 1	0	0	0	1	0	1	0	1	1
	Ophelosia sp. 2	2	0	0	0	2	0	2	0	2
	Ormyromorpha sp. 1	0	0	0	1	0	1	0	1	1
	Paratomicobia? sp. 1	0	0	0	2	0	2	0	2	2
	Pteromalinae sp. 1	0	0	0	1	0	1	0	1	1
	Pteromalinae sp. 2	0	0	0	1	0	1	0	1	1
	Spalangia sp. 1	0	0	0	1	0	1	0	1	1
	Systasis sp. 1	0	2	0	1	0	3	2	1	3

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Pteromalidae spp.	1	84	0	31	1	115	85	31	116
Scelionidae	Baeini sp. 1	0	0	5	1	5	1	0	6	6
	Ceratobaeus nr *ayeshae* Iqbal & Austin, 2000	0	3	0	0	0	3	3	0	3
	Duta sp. 1	0	0	1	0	1	0	0	1	1
	"*Eumicrosoma*" sp. 1	0	0	0	2	0	2	0	2	2
	"*Eumicrosoma*" sp. 2	0	0	1	3	1	3	0	4	4
	"*Eumicrosoma*" sp. 3	0	0	2	12	2	12	0	14	14
	Genatropis sp. 1	5	0	28	0	33	0	5	28	33
	Gryon sp. 1	0	0	37	3	37	3	0	40	40
	Gryon sp. 2	0	0	2	0	2	0	0	2	2
	Idris sp. 1	3	1	1	0	4	1	4	1	5
	Idris sp. 2	0	2	1	1	1	3	2	2	4
	Idris sp. 3	2	0	0	0	2	0	2	0	2
	Idris sp. 4	0	3	0	0	0	3	3	0	3
	Idris sp. 5	0	0	1	0	1	0	0	1	1
	Idris sp. 6	1	0	0	0	1	0	1	0	1
	Idris sp. 7	0	1	0	0	0	1	1	0	1
	Idris sp. 8	0	1	0	0	0	1	1	0	1
	Idris sp. 9	0	0	1	0	1	0	0	1	1
	Idris? sp. 10	0	0	26	4	26	4	0	30	30
	Mirobaeoides ater? (Hickman, 1967)	3	0	0	0	3	0	3	0	3
	Mirobaeoides nr *setosus* Austin, 1986	1	0	0	0	1	0	1	0	1
	Mirobaeoides tasmanicus Dodd, 1914	0	0	1	2	1	2	0	3	3
	Neobaeus sp. 1	0	0	1	0	1	0	0	1	1
	Opisthacantha sp. 1	0	1	9	0	9	1	1	9	10
	Sceliacanthella sp. 1	7	4	22	0	29	4	11	22	33
	Scelio sp. 1	0	0	1	0	1	0	0	1	1
	Scelio sp. 2	3	2	0	0	3	2	5	0	5
	Scelioninae sp. 1	0	0	9	0	9	0	0	9	9
	Telenomus sp. 1	0	1	1	6	1	7	1	7	8

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Telenomus sp. 2	0	0	0	2	0	2	0	2	2
	Trimorus sp. 1	0	0	1	2	1	2	0	3	3
	Trimorus sp. 2	0	0	3	0	3	0	0	3	3
	Trimorus sp. 3	0	0	2	0	2	0	0	2	2
	Trimorus sp. 4	0	0	1	0	1	0	0	1	1
	Trimorus sp. 5	0	0	1	0	1	0	0	1	1
	Trimorus sp. 6	2	0	0	0	2	0	2	0	2
	Trimorus sp. 7	0	0	1	1	1	1	0	2	2
	Trimorus sp. 8	1	0	0	0	1	0	1	0	1
	Trissoclus sp. 1	0	5	1	0	1	5	5	1	6
	Trissoclus sp. 2	0	1	0	1	0	2	1	1	2
	Scelionidae spp.	278	43	481	56	759	99	321	537	858
Signiphoridae	Signiphoridae spp.	0	0	0	2	0	2	0	2	2
Sphecidae	Sphecidae spp.	0	1	0	0	0	1	1	0	1
Tanaostigmatidae	*Tanaostigmodes* sp. 1	0	0	0	1	0	1	0	1	1
	Tanaostigmatidae spp.	0	0	1	1	1	1	0	2	2
Tiphiidae	Tiphiidae spp.	0	0	0	1	0	1	0	1	1
Torymidae	*Megastigmus* sp. 1	0	0	0	1	0	1	0	1	1
	Torymidae spp.	1	9	0	4	1	13	10	4	14
Trichogrammatidae	Trichogrammatidae sp. 1	0	2	0	3	0	5	2	3	5
	Trichogrammatidae sp. 2	0	0	1	0	1	0	0	1	1
	Trichogrammatidae sp. 3	0	1	0	0	0	1	1	0	1
	Trichogrammatidae sp. 4	0	1	0	2	0	3	1	2	3
	Trichogrammatidae sp. 5	0	0	2	0	2	0	0	2	2
	Trichogrammatidae spp.	10	3	14	7	24	1	13	21	34
Vespidae	*Paralastor* sp. 1	0	0	0	1	0	1	0	1	1
	Vespula germanica (Fabricius, 1793)	0	0	5	0	5	0	0	5	5
Total Hymenoptera		3396	1308	1885	863	3346	1721	4704	2748	7452

Lepidoptera (Peter McQuillan)

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
Anthelidae	*Anthela* sp. 1 (L)	0	0	1	1	1	1	0	2	2
	Pterolocera sp. 1 (L)	1	8	0	0	1	8	9	0	9
Arctiidae	*Damais procrena* (Meyrick, 1886)	0	2	0	1	0	3	2	1	3
	Lithosiinae sp. 1 (L)	1	4	3	7	4	11	5	10	15
	Phaos interfixa Walker, 1856 (L)	0	0	2	0	2	0	0	2	2
	Arctiidae indet. (L)	0	0	1	0	1	0	0	1	1
Batrachedidae	Batrachedidae sp. 1	0	0	1	0	1	0	0	1	1
Cosmopterygidae	Cosmopterygidae indet.	1	6	0	6	1	12	7	6	13
Elachistidae	Elachistidae sp.	2	20	1	31	3	51	22	32	54
Gelechiidae	Gelechiidae sp. 1	1	0	0	0	1	0	1	0	1
Gelechioidea	Gelechioidea (L)	0	6	0	11	0	17	6	11	17
Geometridae	*Anachloris* sp. 1	0	8	0	0	0	8	8	0	8
	*Aponotoreas epicrossa (*Meyrick, 1891)	1	16	0	6	1	22	17	6	23
	Chlorocoma rhodothrix Turner, 1922	0	1	0	0	0	1	1	0	1
	Chrysolarentia bertha (Swinhoe, 1902)	0	2	1	0	1	2	2	1	3
	Chrysolarentia chrysocyma (Meyrick, 1891)	0	0	0	1	0	1	0	1	1
	Chrysolarentia indet.	0	8	0	6	0	14	8	6	14
	Dichromodes sp. 1	0	7	0	2	0	9	7	2	9
	Ennominae sp. 1 (L)	0	0	0	1	0	1	0	1	1
	Ennominae sp. 2 (L)	0	42	0	42	0	84	42	42	84
	Ennominae sp. 5 (L)	0	3	0	2	0	5	3	2	5
	Ennominae sp. 6 (L)	0	10	0	0	0	1	10	0	10
	Ennominae sp. 7 (L)	0	1	0	0	0	1	1	0	1
	Ennominae sp. 8 (L)	1	10	0	7	1	17	11	7	18
	Ennominae sp. 9 (L)	0	1	0	0	0	1	1	0	1
	Geometridae sp. 1 (L)	0	16	0	0	0	16	16	0	16
	Geometridae sp. 2 (L)	0	2	0	0	0	2	2	0	2
	Geometridae sp. 3 (L)	0	0	2	4	2	4	0	6	6
	Geometridae sp. 4 (L)	1	1	0	0	1	1	2	0	2
	Geometridae sp. 7 (L)	2	2	0	1	2	3	4	1	5

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Geometridae sp. 8 (L)	0	6	0	0	0	6	6	0	6
	Larentiinae indet.	0	3	0	1	0	4	3	1	4
	Neritodes verrucata Guenée, 1857)	0	1	0	0	0	1	1	0	1
	Oenochrominae sp. 1 (L)	1	5	0	0	1	5	6	0	6
	Geometridae indet.	0	8	0	1	0	9	8	1	9
	Geometridae indet. (L)	8	30	1	3	9	33	38	4	42
Glyphipiterygidae	*Glyphipterix* sp. 1	0	10	0	3	0	13	10	3	13
Heliozelidae sp. 1 (A)	Heliozelidae sp. 1	1	6	1	0	2	6	7	1	8
Hesperiidae	*Hesperilla donnysa* Hewitson, 1868	0	0	0	1	0	1	0	1	1
Hypertrophidae	*Thudaca* sp. 1	0	9	0	0	0	9	9	0	9
Lycaenidae	*Neolucia hobartensis* (Miskin, 1890)	0	99	0	2	0	11	99	2	101
Nepticulidae	Nepticulidae indent.	0	2	0	0	0	2	2	0	2
Noctuidae	Acronyctinae sp. 1 (L)	1	0	1	0	2	0	1	1	2
	Agrotis sp. 1 (L)	0	0	1	0	1	0	0	1	1
	Hadeninae sp. 1 (L)	0	0	2	0	2	0	0	2	2
	Persectania sp. 1	0	0	0	12	0	12	0	12	12
	Noctuidae indet. (L)	0	0	3	1	3	1	0	4	4
Nolidae	*Nola* sp. 1 (L)	0	14	0	0	0	14	14	0	14
Nymphalidae	*Argynnina tasmanica* (Lyell, 1900) (L)	0	0	0	1	0	1	0	1	1
	Geitoneura klugii (Guérin-Méneville, 1930)	0	0	1	1	1	1	0	2	2
	Heteronympha cordacae (Geyer, 1832)	0	0	0	2	0	2	0	2	2
	Oreixenica lathoniella (Westwood, 1851)	0	0	0	2	0	2	0	2	2
Oecophoridae	*Barea* sp. 1 (L)	1	0	0	0	1	0	1	0	1
	Macronemata sp. 1	0	7	0	2	0	9	7	2	9
	Oecophoridae sp. 1 (L)	0	90	0	4	0	94	90	4	94
	Oecophoridae sp. 10 (L)	0	1	0	0	0	1	1	0	1
	Oecophoridae sp. 12 (L)	0	0	0	1	0	1	0	1	1
	Oecophoridae sp. 14 (L)	0	0	0	1	0	1	0	1	1
	Oecophoridae sp. 15 (L)	0	1	0	8	0	9	1	8	9
	Oecophoridae sp. 16 (L)	0	38	1	0	1	38	38	1	39
	Oecophoridae sp. 17 (L)	1	0	0	0	1	0	1	0	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Oecophoridae sp. 2 (L)	0	39	0	5	0	44	39	5	44
	Oecophoridae sp. 3 (L)	1	18	0	90	1	18	19	90	109
	Oecophoridae sp. 4 (L)	0	1	0	0	0	1	1	0	1
	Oecophoridae sp. 5 (L)	0	11	1	22	1	33	11	23	34
	Oecophoridae sp. 6 (L)	0	1	0	0	0	1	1	0	1
	Oecophoridae sp. 7 (L)	0	0	0	231	0	231	0	231	231
	Oecophoridae sp. 8 (L)	0	0	0	1	0	1	0	1	1
	Stathmopoda sp. 1 (L)	0	2	0	0	0	2	2	0	2
	Oecophoridae indet.	0	24	2	3	2	27	24	5	29
	Oecophoridae indet. (L)	1	22	0	4	1	26	23	4	27
Psychidae	*Lepidoscia* sp. 1 (L)	0	0	1	4	1	4	0	5	5
	Narycia sp. 1 (L)	0	0	10	3	1	3	0	13	13
	Narycia sp. 2 (L)	0	1	0	0	0	1	1	0	1
	Narycia sp. 4 (L)	0	2	0	2	0	4	2	2	4
	Plutorectis sp. 1	0	0	1	0	1	0	0	1	1
Pterophoridae	Pterophoridae sp. 1	0	0	0	1	0	1	0	1	1
Pyralidae	*Hednota* sp. 1	1	0	0	0	1	0	1	0	1
	Heliothela sp. 1	0	0	0	2	0	2	0	2	2
	Pyralidae sp. 1	0	0	0	0	0	0	0	0	0
	Pyralidae sp. 2	0	0	0	0	0	0	0	0	0
	Tipaena patulella Walker, 1864	0	0	0	1	0	1	0	1	1
Thaumetopoeidae	*Epicoma* sp. 1 (L)	0	1	0	0	0	1	1	0	1
Tineidae	Tineidae sp. 1	0	0	3	0	3	0	0	3	3
Tortricidae	*Epiphyas* sp. 1	0	6	0	0	0	6	6	0	6
	Ericodesma sp. 1	0	0	0	2	0	2	0	2	2
	Isochorista sp. 1 (L)	0	1	0	2	0	3	1	2	3
	Spilonota sp. 1 (L)	0	1	0	0	0	1	1	0	1
	Tortricidae indet.	0	3	0	1	0	4	3	1	4
	Tortricidae indet. (L)	0	1	0	9	0	1	1	9	10
Zygaenidae	Zygaenidae sp. 1 (L)	0	29	0	0	0	29	29	0	29

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Zygaenidae sp. 2	0	13	0	0	0	13	13	0	13
Lepidoptera indet. (A)	Lepidoptera indet.	1	2	0	1	1	3	3	1	4
Lepidoptera indet. (L)	Lepidoptera indet. (L)	8	5	0	2	8	7	13	2	15
Total Lepidoptera		36	689	41	561	68	1052	725	602	1327
Mecoptera (Gunther Theischinger)										
Apteropanorpidae	*Apteropanorpa warra* Palmer, Trueman & Yeates, 2007	0	0	1	0	1	0	0	1	1
Nannochoristidae	*Nannochorista* sp. 1	1	0	0	0	1	0	1	0	1
Total Mecoptera		1	0	1	0	2	0	1	1	2
Neuroptera (Tim New)										
Coniopterygidae	Coniopterygidae indet. (L)	0	4	0	1	0	5	4	1	5
Odonata (Gunther Theischinger & Michael Driessen)										
Coenagrionidae	*Ischnura aurora* (Brauer, 1865)	0	3	0	2	0	5	3	2	5
Lestidae	*Austrolestes analis* (Rambur, 1842)	0	0	0	5	0	5	0	5	5
	Austrolestes annulosus (Selys, 1862)	0	3	0	0	0	3	3	0	3
	Austrolestes psyche (Hagen, 1862)	0	5	0	8	0	13	5	8	13
Synthemistidae	*Synthemis tasmanica* Tillyard, 1910	0	0	0	1	0	1	0	1	1
Total Odonata		0	11	0	16	0	27	11	16	27
Orthoptera (Michael Driessen)										
Caelifera										
Acrididae	*Phaulacridium vittatum* (Sjöstedt, 1920)	3	3	4	6	7	9	6	10	16
	Russalpia albertisi (Bolivar, 1898)	0	0	5	2	5	2	0	7	7
	Russalpia longifurca Key, 1991	58	148	27	97	85	245	206	124	330
	Tasmaniacris tasmaniensis (Bolivar, 1898)	1	0	0	0	1	0	1	0	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
Pyrgophidae	*Monistria concinna* (Walker, 1871)	0	0	1	0	1	0	0	1	1
Tetrigidae	*Tetrix collina* Rehn, 1952	1	0	2	6	3	6	1	8	9
Ensifera		0	0	0	0	0	0	0	0	0
Gryllidae	*Bobilla poene* Otte & Alexander 1983	2117	1	3175	2	5292	3	2118	3177	5295
	Bobilla tasmani Otte & Alexander, 1983	0	0	2	0	2	0	0	2	2
Tettigoniidae	*Conocephalus bilineatus* (Erichson, 1842)	0	3	1	9	1	12	3	10	13
Total Orthoptera		2180	155	3217	122	5397	277	2335	3339	5674
Plecoptera (Gunther Theischinger)										
Gripopterygidae	*Cardioperla divers* McLellan, 1971	0	0	0	1	0	1	0	1	1
	Dinotoperla serricauda Kimmins, 1951	0	0	0	1	0	1	0	1	1
	Leptoperla sp. 1	0	0	1	0	1	0	0	1	1
	Leptoperla varia Kimmins, 1951	0	0	1	1	1	1	0	2	2
Notonemuridae	*Austrocerca tasmanica* (Tillyard, 1924)	0	0	1	1	1	1	0	2	2
	Kimminsoperla williamsi Illies, 1975	0	0	0	32	0	32	0	32	32
Total Plecoptera		0	0	3	36	3	36	0	39	39
Psocoptera (Tim New)										
Caeciliusidae	"*Caecilius*" sp. 1	0	0	0	6	0	6	0	6	6
Psocoptera indet.	Psocoptera indet.	0	3	0	56	0	59	3	56	59
Total Psocoptera		0	3	0	62	0	65	3	62	65
Thysanoptera (Laurence Mound)										
Phlaeothripidae	*Baenothrips moundi* (Stannard, 1970)	0	0	1	4	1	4	0	5	5
	Carientothrips sp. 1	0	2	4	3	4	5	2	7	9
	**Haplothrips driesseni* Mound & Minaei, 2007	0	32	0	10	0	42	32	10	42
	Haplothrips sp. indet.	0	0	0	3	0	3	0	3	3

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Haplothrips victoriensis Bagnall, 1918	0	1	0	0	0	1	1	0	1
	**Minaeithrips driesseni* Mound, 2007	0	26	0	15	0	41	26	15	41
	Phlaeothripidae sp. 1	0	0	0	1	0	1	0	1	1
	Psalidothrips taylori Mound and Walker, 1986	1	9	2	3	3	12	10	5	15
	Zemiathrips sp. 1	0	2	0	1	0	3	2	1	3
	Phlaeothripidae indet. (L)	2	57	4	22	6	79	59	26	85
Thripidae	*Anaphothrips obscurus* (Müller, 1776)	0	0	0	3	0	3	0	3	3
	Chirothrips manicatus (Haliday, 1836)	0	2	0	0	0	2	2	0	2
	Pseudanaphothrips frankstoni? (Steele, 1940)	0	0	0	1	0	1	0	1	1
	Pseudanaphothrips achaetus (Bagnell, 1916)	5	1628	1	877	6	255	1633	878	2511
	Scirtothrips pan? Mound & Walker, 1982	0	1	0	0	0	1	1	0	1
	Thrips australis (Bagnell, 1915)	0	14	0	95	0	19	14	95	109
	Thrips imaginis Bagnell, 1926	2	452	3	479	5	931	454	482	936
	Thrips tabaci Lindemann, 1888	0	1	0	0	0	1	1	0	1
	Thrips wellsae Mound & Masumoto, 2005	0	4	0	16	0	2	4	16	20
	Thripidae indet. (L)	0	65	0	18	0	83	65	18	83
Total Thysanoptera		10	2296	15	1551	25	1489	2306	1566	3872
Trichoptera (Alice Wells and Arturs Neboiss)										
Calamoceratidae	*Anisocentropus* sp. 1	0	1	0	0	0	1	1	0	1
Hydroptilidae	*Oxyethira* "cervinus" sp. nov.	0	2	0	0	0	2	2	0	2
	**Oxyethira driesseni* Wells, 2003	0	0	1	5	1	5	0	6	6
	Tricholeiochiton pennyae Wells 1998	0	20	1	5	1	25	20	6	26
Kokiriidae	*Taskiria austera* Neboiss, 1977	0	5	0	2	0	7	5	2	7
Leptoceridae	*Notalina* sp. 1	0	2	0	3	0	5	2	3	5
	Notoperata maculata (Mosely, 1953)	0	0	0	3	0	3	0	3	3
	Triplectides cuisku (Walker, 1852)	0	15	0	0	0	15	15	0	15
	Triplectides sp. 1	0	4	0	0	0	4	4	0	4
Plectrotarsidae	*Liapota lavara* Neboiss, 1959	2	9	0	5	2	14	11	5	16
	Nanoplectrus trochanasi Neboiss, 1977	0	52	0	0	0	52	52	0	52

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Plectrotarsus gravenhorsti Kolenati, 1848	0	0	0	1	0	1	0	1	1
	Plectrotarsus tasmanicus Mosely, 1936	0	0	0	5	0	5	0	5	5
	Plectrotarsus sp. 1	0	0	0	1	0	1	0	1	1
Polycentropodidae	*Tasmanoplegas spilota* Neboiss, 1977	20	0	160	15	18	15	20	175	195
Total Trichoptera		22	110	162	45	22	155	132	207	339
COLLEMBOLA (Penny Greenslade, Kevin Bonham & Abbey Throssell)										
Entomobryomorpha										
Entomobryidae	*Australotomurus* sp. 1	12	0	0	0	12	0	12	0	12
	Australotomurus sp. 3	2	0	0	0	2	0	2	0	2
	cf. *Drepanura* sp. 1	97	0	33	0	13	0	97	33	130
	Lepidocyrtini sp. 1	0	0	6	0	6	0	0	6	6
	Lepidocyrtus sp. 1	297	1	32	0	329	1	298	32	330
	Sinella sp. 1	2	0	2	0	4	0	2	2	4
	Entomobryidae indet.	13	2	17	12	3	14	15	29	44
Isotomidae	*Acanthomurus* spp.	11367	12	8540	13	1997	25	11379	8553	19932
	cf. *Isotoma* sp. 1	53	0	1	0	54	0	53	1	54
	cf. *Isotoma* sp. 2	179	0	77	0	256	0	179	77	256
	cf. *Tomocerura* sp. 1	204	0	0	0	24	0	204	0	204
	cf. *Tomocerura* sp. 2	11	0	0	0	11	0	11	0	11
	*Cryptopygus antarcticus*Willem, 1901	26	1	64	0	9	1	27	64	91
	Folsomotoma sp. 1	1	0	1	0	2	0	1	1	2
	Parisotoma sp. 1	18	0	0	0	18	0	18	0	18
Paronellidae	*Paronellides* sp. 1	2	0	20	57	22	57	2	77	79
	Paronellides sp. 2	1	1	130	3	131	4	2	133	135
	Paronellides sp. 3	3	9	70	140	73	149	12	210	222
	Paronellides sp. 4	3	22	306	0	39	22	25	306	331
	Paronellides sp. 5	32	0	255	0	287	0	32	255	287
	Paronellidae indet.	5	1	347	5	352	6	6	352	358

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
Tomoceridae	*Lasofinius* spp.	263	0	329	0	592	0	263	329	592
	Lepidophorella sp. 1	4	0	242	0	246	0	4	242	246
	Lepidosira australica (Schött, 1917)	1	0	0	0	1	0	1	0	1
	Novacerus sp. 1	3	0	0	0	3	0	3	0	3
Neelipleona										
Neelidae	*Neelides* spp.	3	0	20	0	23	0	3	20	23
Poduromorpha										
Brachystomellidae	*Brachystomella* sp. 1	14	0	24	0	38	0	14	24	38
	Cassagnella sp. 1	11	2	26	2	37	4	13	28	41
	Setanodosa sp. 1	0	23	0	5	0	28	23	5	28
	Brachystomellidae indet.	4	0	307	0	311	0	4	307	311
Hypogastruridae	*Ceratophysella* sp. 1	8	0	13	0	21	0	8	13	21
	Hypogastrura purpurescens (Lubbock, 1868)	0	0	458	0	458	0	0	458	458
	Hypogastruridae indet.	0	0	1	0	1	0	0	1	1
Neanuridae	*Acanthanura* cf. *bicornis* (Womersley, 1940)	0	0	4	0	4	0	0	4	4
	Acanthanura cf. *dendyi* (Lubbock, 1899)	2	0	41	0	43	0	2	41	43
	Acanthanura spp.	16	0	82	0	98	0	16	82	98
	Ceratrimeria sp. 1	0	0	3	0	3	0	0	3	3
	Friesea sp. 1	3	0	0	0	3	0	3	0	3
	nr *Friesea* sp. 1	1	0	49	0	5	0	1	49	50
	Neanuridae indet.	53	0	143	1	196	1	53	144	197
Odontellidae	Odontellidae indet.	1300	0	2588	0	3888	0	1300	2588	3888
	Odontellidae sp. 1	0	0	207	0	207	0	0	207	207
	Odontellidae sp. 2	377	0	0	0	377	0	377	0	377
Poduroidea indet.		61	0	31	0	92	0	61	31	92
Symphypleona										
Bourletiellidae	*Corynephoria* sp. 1	1168	943	1640	2146	288	389	2111	3786	5897
	Rastriopes sp. 1	81	1070	1	3	82	173	1151	4	1155

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
Dicyrtomidae	Dicyrtomidae indet.	54	2	302	0	356	2	56	302	358
Katiannidae	cf. *Parakatianna* sp. 3	0	0	0	3	0	3	0	3	3
	Katianna sp. 1	92	0	224	0	316	0	92	224	316
	Katianna sp. 2	11	0	36	0	47	0	11	36	47
	Katiannidae indet	742	55	1921	0	2663	55	797	1921	2718
	Katianninae sp. 1	0	0	108	0	18	0	0	108	108
	Parakatianna sp. 1	201	3839	7	15	28	3854	4040	22	4062
	Parakatianna sp. 2	943	188	513	0	1456	188	1131	513	1644
	Polykatianna cf. *aurea* (Womersley, 1932)	1123	4392	1212	1282	2335	5674	5515	2494	8009
	Sminthurinus sp. 1	143	0	0	0	143	0	143	0	143
	Sminthurinus sp. 2	743	0	393	0	1136	0	743	393	1136
	Sminthurinus spp.	105	0	1	1	16	1	105	2	107
	Sminthurididae indet.	124	0	248	0	372	0	124	248	372
	Sminthurides sp. 1	119	0	207	0	326	0	119	207	326
	Sphaeridia sp. 1	5	0	41	0	46	0	5	41	46
Collembola indet.	Collembola indet.	122	0	30	0	152	0	122	30	152
Total Collembola		20228	10563	21353	3688	20071	10651	30791	25041	55832
CHILOPODA (Bob Mesibov)										
Lithobiomorpha										
Henicopidae	*Henicops maculatus* Newport, 1845	17		48		65		17	48	65
Total Chilopoda		17		48		65		17	48	65
DIPLOPODA (Bob Mesibov)										
Chordeumatida										
Metopidiotrichidae	*Australeuma jeekeli* Golovatch, 1986	2	0	0	0	2	0	2	0	2
	Australeuma simile Golovatch, 1986	55	0	346	0	41	0	55	346	401

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
Polydesmida										
Dalodesmidae	*Dasystigma margaretae* (Jeekel, 1984)	0	0	8	0	8	0	0	8	8
	Dasystigma tyleri Mesibov, 2003	2	0	2	0	4	0	2	2	4
	Gasterogramma imber Mesibov, 2003	1	0	0	0	1	0	1	0	1
	Gasterogramma psi Jeekel, 1982	0	0	4	0	4	0	0	4	4
	Gasterogramma sp. 1	0	0	11	0	11	0	0	11	11
	Lissodesmus cornutus Mesibov, 2006	1	0	0	0	1	0	1	0	1
Dalodesmidea family uncertain	*Paredrodesmus monticolus* Mesibov, 2003	0	0	118	0	118	0	0	118	118
Spirostreptida										
Lulomorphidae	*Amastigogonus* sp. 1	0	0	5	0	5	0	0	5	5
Total Diplopoda		61	0	494	0	195	0	61	494	555
SYMPHYLA (Michael Driessen)										
Scutigerellidae	*Hanseniella* spp.	1	0	1	0	2	0	1	1	2
MOLLUSCA										
GASTROPODA (Kevin Bonham)										
Eupulmonata										
Arionidae	*Arion intermedius* Normand, 1852	0	0	5	0	5	0	0	5	5
Charopidae	*Allocharopa legrandi* (Cox, 1868)	0	0	1	0	1	0	0	1	1
	Mulathena fordei (Brazier, 1871)	1	0	0	0	1	0	1	0	1
	Pernagera kingstonensis (Legrand, 1871)	0	0	3	0	3	0	0	3	3
	Stenacapha cf. *vitrinaformis* Legrand, 1871	125	0	19	0	144	0	125	19	144
	Stenacapha hamiltoni (Cox, 1868)	0	0	13	0	13	0	0	13	13
	Thryasona sp. "Wedge"	11	0	0	0	11	0	11	0	11
	Thryasona sp. nov.	3	0	0	0	3	0	3	0	3
Helicarionidae	*Helicarion cuvieri* Férussac, 1921	8	0	33	0	41	0	8	33	41
Punctidae	*Paralaoma caputspinulae* (Reeve, 1851)	0	0	1	0	1	0	0	1	1

Group	Taxa	Lowland		Montane		Totals				
		PF	SW	PF	SW	PF	SW	Low.	Mon.	Grand
	Paralaoma cf. *halli* (Legrand, 1870)	0	0	4	0	4	0	0	4	4
Total Gastropoda		148	0	79	0	227	0	148	79	227
PLATYHELMINTHES										
TURBELLARIA (Leigh Winsor)										
Tricladida										
Geoplanidae	*Tasmanoplana tasmaniana*? (Darwin, 1844)	0	0	15	0	15	0	0	15	15
	Artioposthia sp. 1	0	0	5	0	5	0	0	5	5
	Artioposthia sp. 5	22	0	67	0	89	0	22	67	89
	Caenoplaninae sp. 1	3	0	10	0	13	0	3	10	13
	Caenoplaninae sp. 2	0	0	3	0	3	0	0	3	3
	Caenoplaninae sp. 3	0	0	5	0	5	0	0	5	5
Paludicola	Paludicola sp. 1	0	0	8	0	8	0	0	8	8
Total Tricladida		25	0	113	0	138	0	25	113	138
Total		31130	42588	36771	30736	41729	57322	73718	67507	141090

Appendix 2 Groups of taxa with similar monthly distributions

Groups of taxa with similar monthly distribution patterns based on cluster analysis.

Sample sizes for each taxon at the montane and lowland moorland locations are shown.

Only taxa with sample size ≥30 at a location were used in cluster analysis.

'+' = species present at location (<30),'-' = species not recorded at location, A–E = group where taxon for that location was clustered and sample size given.

M = sampling method; pitfall trap (P) and sweep net (S).

Order	Family	Species	Lowland	Montane	M
Group A					
Acarina	Anystidae	*Walzia australica* Womersley	34	397	S
Acarina	Bdellidae	*Bdellodes* spp.	+	52	P
Acarina	Erythraeidae	*Wartookia rebeccae* Southcott	67	97	S
Acarina	Eupodidae	nr *Eupodes* sp. nov. 1	-	256	P
Acarina	Liebstadiidae	*Reductobates humeratus* Balogh and Mahunka	+	99	S
Acarina	Parakalummatidae	*Sandenia rotunda* Wallwork	2151	132	S
Acarina	Penthalaeidae	*Halotydeus* sp. 1	+	62	P
Acarina	Smarididae	*Hirstiosoma tasmaniensis* Womersley and Southcott	+	57	P
Amphipoda	Talitridae	*Keratroides vulgaris* (Friend)	+	110	P
Araneae	Amaurobiidae	Amaurobiidae sp. 2	+	30	P
Araneae	Araneidae	*Araneus acuminatus* (L. Koch)	+	230	S
Araneae	Araneidae	*Araneus* sp. 2	B	102	S
Araneae	Araneidae	*Eriophora pustulosa* (Walckenaer)	C	360	S

Order	Family	Species	Lowland	Montane	M
Araneae	Clubionidae	*Cheiracanthium* sp. 1	32	+	S
Araneae	Clubionidae	*Clubiona* sp. 4	+	50	S
Araneae	Linyphiidae	Linyphiidae sp. 3	+	33	S
Araneae	Lycosidae	Lycosidae sp. 7	-	66	P
Araneae	Tetragnathidae	*Tetragnatha* sp. 1	90	680	S
Araneae	Thomisidae	*Diaea rosea* (L. Koch)	325	33	S
Coleoptera	Scirtidae	*Pseudomicrocara* sp. 2	-	96	S
Coleoptera	Staphylinidae	Aleochaerinae spp.	+	31	P
Collembola	Bourletiellidae	*Corynephoria* sp. 1	357	469	P&S
Collembola	Bourletiellidae	*Rastriopes* sp. 1	143	-	S
Collembola	Brachystomellidae	indet.	+	307	P
Collembola	Dicyrtomidae	indet.	C	212	P
Collembola	Istomidae	*Acanthomurus* spp.	4850	4766	P
Collembola	Katiannidae	indet. imm.	646	1843	P&S
Collembola	Katiannidae	Katianninae n. gen.	-	108	P
Collembola	Katiannidae	*Polykatianna* cf. *aurea* (Womersley)	C	1934	P&S
Collembola	Neanuridae	*Acanthanura* cf. *dendyi* (Lubbock)	+	41	P
Collembola	Neanuridae	Indet. imm.	C	143	P
Diptera	Cecidomyiidae	Cecidomyiidae sp. 3	237	47	P
Diptera	Ceratopogonidae	Ceratopogonidae sp. 13	43	265	S
Diptera	Chironomidae	Chironomidae sp. 15	77	163	S
Diptera	Chironomidae	Chironomidae sp. 8	-	93	S
Diptera	Empididae	Empididae sp. 14	-	146	S
Diptera	Empididae	*Hilaropus* sp. 1	+	34	S
Diptera	Helosciomyzidae	Helosciomyzidae sp. 5	-	102	S

Order	Family	Species	Lowland	Montane	M
Diptera	Muscidae	Muscidae sp. 3	-	786	S
Diptera	Phoridae	Phoridae sp. 3	-	61	S
Diptera	Phoridae	Phoridae sp. 5	136	150	P
Diptera	Sciaridae	Sciaridae sp. 1	-	59	P
Diptera	Tipulidae	*Molophilus flavonotatus* Skuse	B	33	S
Hemiptera	Acanthosomatidae	Acanthosomatidae sp. 1	54	+	S
Hemiptera	Cicadellidae	Ulopinae sp. 2	155	250	S
Hemiptera	Cicadellidae	Ulopinae sp. 3	578	+	S
Hemiptera	Hemiptera	indet. nymph	156	E	P&S
Hemiptera	Lygaeidae	*Nysius* sp. 1	+	31	S
Hemiptera	Miridae	Miridae sp. 4	+	30	S
Hemiptera	Pachygronthidae	Pachygronthidae sp. 1	+	671	S
Hymenoptera	Braconidae	Rogadinae sp. 3	+	32	S
Hymenoptera	Formicidae	*Anonychomyrma* sp. 1	1106	+	P
Hymenoptera	Scelionidae	*Gryon* sp. 1	-	37	P
Lepidoptera	Oecophoridae	Oecophoridae sp. 7	-	231	S
Orthoptera	Gryllidae	*Bobilla poene* Otte and Alexander	1233	911	P
Polydesmida	indet. family	*Paredrodesmus monticolus* Mesibov	-	95	P
Thysanoptera	Thripidae	*Pseudanaphothrips achaetus* (Bagnall)	197	54	S
Thysanoptera	Thripidae	*Thrips australis* (Bagnall)	-	56	S
Tricladida	Geoplanidae	*Artioposthia* sp. 5	-	35	P
Group B					
Araneae	Araneidae	*Araneus* sp. 2	39	A	S
Araneae	Araneidae	*Cyclosa* sp. 1	53	+	S
Araneae	Dictynidae	Dictynidae sp. 1	C	47	S

Order	Family	Species	Lowland	Montane	M
Coleoptera	Lathridiidae	*Corticaria* sp. 1	+	130	S
Collembola	Entomobryidae	cf. *Drepanura* sp. 1	57	+	P
Collembola	Odontellidae	indet. imm.	C	593	P
Diptera	Chironomidae	Chironomidae sp. 18	+	294	S
Diptera	Chironomidae	Chironomidae sp. 28	98	C	S
Diptera	Tipulidae	*Limonia* sp. 2	+	31	S
Diptera	Tipulidae	*Molophilus flavonotatus* Skuse	80	A	S
Hemiptera	Psyllidae	Psyllidae sp. 2	44	+	S
Psocoptera	indet. family	nymph	+	56	S
Group C					
Acarina	Eriorhynchidae	*Eriorhynchus ramosus* Qin & Halliday	+	30	P
Araneae	Agelenidae	Agelenidae sp. 1	128	+	S
Araneae	Amaurobiidae	*Tasmarubrius hickmani* Davies	+	170	P
Araneae	Araneidae	*Araneus* sp. 12	+	39	S
Araneae	Araneidae	*Eriophora pustulosa* (Walckenear)	30	A	S
Araneae	Dictynidae	Dictynidae sp. 1	101	B	S
Araneae	Theridiidae	*Dipoena* sp. 3	51	-	S
Collembola	Dicyrtomidae	indet.	45	A	P
Collembola	Entomobryidae	*Lepidocyrtus* sp. 1	97	+	P
Collembola	Istomidae	cf. *Isotoma* sp. 1	53	+	P
Collembola	Istomidae	cf. *Isotoma* sp. 2	+	36	P
Collembola	Istomidae	*Cryptopygus antarcticus* Willem	+	45	P
Collembola	Katiannidae	*Parakatianna* sp. 1	3839	+	S
Collembola	Katiannidae	*Parakatianna* sp. 2	943	513	P

Order	Family	Species	Lowland	Montane	M
Collembola	Katiannidae	*Polykatianna* cf. *aurea* (Womersley)	2766	A	P&S
Collembola	Katiannidae	*Sminthurinus* spp.	105	+	P
Collembola	Neanuridae	indet. imm.	52	A	P
Collembola	Odontellidae	indet. imm.	30	B	P
Collembola	Paronellidae	indet.imm.	+	333	P
Collembola	Paronellidae	*Paronellides* sp. 1	+	57	S
Collembola	Paronellidae	*Paronellides* sp. 2	+	76	P
Collembola	Paronellidae	*Paronellides* sp. 3	+	138	S
Collembola	Paronellidae	*Paronellides* sp. 4	-	264	P
Collembola	Paronellidae	*Paronellides* sp. 5	-	35	P
Collembola	Tomoceridae	*Lasofinius* spp.	263	329	P
Diptera	Anisopodidae	Anisopodidae sp. 1	199	E	P
Diptera	Chironomidae	Chironomidae sp. 16	420	E	S
Diptera	Chironomidae	Chironomidae sp. 17	+	145	S
Diptera	Chironomidae	Chironomidae sp. 23	D	96	S
Diptera	Chironomidae	Chironomidae sp. 28	B	1169	S
Diptera	Dolichopodidae	*Chrysotimus* sp. 1	88	+	S
Diptera	Mycetophilidae	Mycetophilidae sp. 1	+	132	P
Hemiptera	Sternorrhyncha	indet.	156	132	P&S
Lepidoptera	Geometridae	Ennominae sp. 2	42	42	S
Lepidoptera	Oecophoridae	Oecophoridae sp. 2	37	+	S
Group D					
Araneae	Lycosidae	*Pardosa* sp. 1	37	E	P
Coleoptera	Carabidae	*Scopodes* spp.	+	47	P
Coleoptera	Curculionidae	*Desiantha* sp. 1	+	34	S

Order	Family	Species	Lowland	Montane	M
Collembola	Istomidae	cf. *Tomocerura* sp. 1	204	-	P
Diptera	Chironomidae	Chironomidae sp. 16	E	481	S
Diptera	Chironomidae	Chironomidae sp. 23	286	C	S
Diptera	Chironomidae	Chironomidae sp. 4	-	141	S
Diptera	Dolichopodidae	*Chrysotus parapicalis* (Bickel & Dyte)	58	32	S
Diptera	Empididae	?*Iteaphila* sp. 1	206	-	S
Diptera	Psychodidae	Psychodidae sp. 1	+	190	P
Diptera	Sphaeroceridae	Sphaeroceridae sp. 1	+	363	P
Hymenoptera	Formicidae	*Iridomyrmex* sp. 1	1366	367	P
Group E					
Acarina	Oribatellidae	*Safrobates miniporus* Mahunka	-	140	P
Araneae	Araneidae	*Araneus* sp. 19	+	42	S
Araneae	Clubionidae	*Clubiona* sp. 1	+	30	S
Araneae	Lycosidae	*Pardosa* sp. 1	D	44	P
Chordeumatida	Metopidiotrichidae	*Australeuma simile* Golovatch	+	165	P
Coleoptera	Scirtidae	*Cyphon* sp. 1	-	33	S
Coleoptera	Scydmaenidae	*Euconnus* sp. 1	-	35	P
Collembola	Entomobryidae	indet. imm.	+	39	P&S
Collembola	Katiannidae	*Katianna* sp. 2	+	36	P
Collembola	Sminthuridae	*Sminthurides* sp. 1	-	40	P
Collembola	Sminthuridae	*Sphaeridia* sp. 1	+	41	P
Diptera	Anisopodidae	Anisopodidae sp. 1	30	C	P
Diptera	Ceratopogonidae	Ceratopogonidae sp. 16	+	58	S
Diptera	Chironomidae	Chironomidae sp. 24	49	+	S

Order	Family	Species	Lowland	Montane	M
Diptera	Chironomidae	Chironomidae sp. 16	+	53	S
Diptera	Chironomidae	Chironomidae sp. 9	-	180	S
Diptera	Tipulidae	*Diemenomyia* sp. 1	-	173	S
Diptera	Empididae	Empididae sp. 4	-	40	S
Diptera	Muscidae	Muscidae sp. 2	+	118	S
Diptera	Simuliidae	Simuliidae sp. 1	56	+	S
Hemiptera	Cercopidae	Cercopidae sp. 1	-	166	S
Hemiptera	Hemiptera	indet. nymph	A	168	P&S
Lepidoptera	Lycaenidae	*Neolucia hobartensis* (Miskin)	95	-	S
Lepidoptera	Oecophoridae	Oecophoridae sp. 1	90	+	S
Plecoptera	Notonemuridae	*Kimminsoperla williamsi* Illies	-	32	S

Appendix 3 Predictor variables for higher taxonomic surrogates

Variables used to predict effectiveness of higher taxonomic surrogates

Taxa: first letter: location: lowland (L) and montane (m); second letter: method: pitfall (P) or sweep (S); 3-5th letters = taxonomic group: Aca = Acarina, Amp =Amphipoda, Ara – Araneae, Chi = Chilopoda, Cop = Coleoptera, Com = Collembola, Dec = Decapoda, Gas = Gastropoda, Hem = Hemiptera, Iso = Isoptera, Lep = Lepidoptera, Opi = Opilionida, Ort = Orthoptera, Thy = Thysanoptera, Trh = Trichoptera; 6th letter = family (F) or order (O) data set. No = number of, Ind = individuals, Sp = species, Com = commonest species, Fam = family, HigherT = higher taxa (family or order), SD = standard deviation, CV = coefficient of variation, Rho = spearman rank coefficient between species matrices and higher taxonomic (family or order) matrices.

Taxa	NoInd	NoSp	NoCom	NoFam	NoOrd	Sp/HigherT	Ind/Sp	SDInd/Sp	CVIndSP	%Com	SDSp/Fam	CVSp/Fam	SDInd/Fam	CVInd/Fam	Rho
LPAcaF	682	62	106	30	1	2.07	11.00	23.37	212.47	15.54	1.60	77.21	35.16	154.66	0.93
LPAmpF	109	5	97	2	1	2.50	21.80	42.14	193.30	88.99	2.12	84.80	74.25	136.23	0.83
LPAraF	743	76	111	27	1	2.81	9.78	16.68	170.65	14.94	2.79	99.04	44.78	162.72	0.64
LPChiF	65	2	51	1	1	2.00	32.50	26.16	80.50	78.46	0.00	0.00	26.16	80.50	0.91
LPCopF	147	47	19	18	1	2.61	3.13	3.89	124.28	12.93	2.53	99.92	11.17	129.22	0.53
LPComF	10005	27	6529	14	1	1.93	370.56	1251.91	337.85	65.26	1.27	65.79	1784.95	249.76	0.92
LPDecF	25	5	15	1	1	5.00	5.00	5.83	116.62	60.00	0.00	0.00	0.00	0.00	0.54
LPGasF	135	4	116	2	1	2.00	33.75	54.99	162.92	85.93	1.41	70.70	85.56	126.76	0.82
LPHemF	215	30	77	14	1	2.14	7.17	15.09	210.60	35.81	2.77	129.26	23.39	152.30	0.89
LPIsoF	36	2	31	1	1	2.00	18.00	18.39	102.14	86.11	0.00	0.00	0.00	0.00	0.87
LPOpiF	77	5	63	2	1	2.50	15.40	26.74	173.61	81.82	2.12	84.84	34.65	89.99	1.00
LPOrtF	927	3	884	3	1	1.00	309.00	498.39	161.29	95.36	0.00	0.00	498.39	161.29	1.00
LSAcaF	4626	28	1924	17	1	1.65	165.21	426.70	258.27	41.59	1.50	90.95	701.17	257.67	0.97
LSAraF	1889	88	535	18	1	4.89	21.47	70.36	327.78	28.32	6.04	123.50	156.33	148.96	0.92
LSCopF	1340	52	437	23	1	2.26	25.77	84.12	326.44	32.61	2.42	102.41	178.37	292.85	0.81
LSComF	3789	7	2152	5	1	1.40	541.29	804.69	148.66	56.80	0.55	39.14	1037.01	136.84	0.73
LSHemF	2349	52	597	14	1	3.71	45.17	120.32	266.36	25.42	4.58	123.34	318.78	189.99	0.82

Taxa	NoInd	NoSp	NoCom	NoFam	NoOrd	Sp/HigherT	Ind/Sp	SDInd/Sp	CVIndSP	%Com	SDSp/Fam	CVSp/Fam	SDInd/Fam	CVInd/Fam	Rho
LSLepF	250	34	38	18	1	1.89	7.35	8.55	116.31	15.20	3.40	130.17	24.99	129.97	0.82
LSOrtF	125	3	122	2	1	1.50	41.67	69.57	166.98	97.60	0.00	0.00	69.57	166.97	1.00
LSThyF	1934	9	1431	3	1	3.00	214.89	477.53	222.22	73.99	0.71	15.71	1295.42	133.96	0.86
LSTrhF	88	7	52	4	1	1.75	12.57	18.03	143.39	59.09	0.96	54.69	27.40	124.54	0.71
MPAraF	1064	90	271	28	1	3.21	11.82	34.94	295.55	25.47	2.97	92.40	90.62	238.47	0.72
MPCopF	371	59	84	16	1	3.69	6.29	12.87	204.67	22.64	3.02	81.90	29.36	126.62	0.75
MPAcaF	1099	72	114	31	1	2.32	15.26	26.81	175.64	10.37	2.61	112.38	58.51	165.04	0.89
MPComF	7567	28	3786	14	1	2.00	270.25	728.16	269.44	50.03	1.18	59.00	1009.66	186.80	0.93
MPHemF	410	40	223	18	1	2.22	10.25	35.17	343.12	54.39	3.51	157.95	73.13	321.06	0.80
MPTrhF	160	1	160	1	1	1.00	160.00		0.00	100.00		0.00		0.00	1.00
MPAmpF	381	4	303	2	1	2.00	95.25	142.94	150.07	79.53	1.41	70.50	267.99	140.68	0.69
MPTrlF	54	7	32	2	1	3.50	7.71	10.93	141.69	59.26	3.54	101.14	26.87	99.52	0.69
MPGasF	65	6	28	4	1	1.50	10.83	10.30	95.08	43.08	1.00	66.67	16.01	98.52	0.84
MPChiF	213	2	181	2	1	1.00	106.50	105.36	98.93	84.98	0.00	0.00	105.36	98.93	1.00
MPDipF	45	5	23	2	1	2.50	9.00	8.37	93.00	51.11	2.12	84.80	0.71	3.16	0.80
MPOrtF	2279	5	2264	4	1	1.25	455.80	1010.82	221.77	99.34	1.00	80.00	1129.18	198.19	1.00
MPIsoF	34	3	29	2	1	1.50	11.33	15.37	135.62	85.29	0.71	47.33	22.63	133.12	0.79
MPOpiF	44	5	25	2	1	2.50	8.80	10.26	116.59	56.82	2.12	84.80	12.73	57.86	0.82
MPLepF	27	14	9	10	1	1.40	1.93	2.16	112.00	33.33	0.97	69.29	2.91	107.78	0.71
MSAraF	1631	82	289	17	1	4.82	19.89	51.58	259.32	17.72	5.51	114.23	134.15	139.83	0.79
MSCopF	402	45	138	23	1	1.96	8.93	21.64	242.24	34.33	2.01	102.73	41.81	219.48	0.83
MSAcaF	2509	32	1075	18	1	1.78	78.41	224.22	285.97	42.85	1.63	91.69	324.88	233.07	0.98
MSComF	2178	6	1845	5	1	1.20	363.00	737.17	203.08	84.71	0.45	37.50	801.57	184.02	1.00
MSHemF	2602	81	857	16	1	5.06	32.12	126.99	395.32	32.94	8.40	165.93	279.60	171.93	0.84
MSThyF	1350	10	823	3	1	3.33	135.00	280.37	207.68	60.96	0.00	0.00	927.72	137.44	0.83
MSOrtF	100	6	80	4	1	1.50	16.67	31.08	186.48	80.00	1.00	66.67	42.03	168.12	0.90

Taxa	NoInd	NoSp	NoCom	NoFam	NoOrd	Sp/HigherT	Ind/Sp	SDInd/Sp	CVIndSP	%Com	SDSp/Fam	CVSp/Fam	SDInd/Fam	CVInd/Fam	Rho
MSLepF	191	22	81	12	1	1.83	8.68	17.83	205.37	42.41	1.75	95.45	32.25	202.62	0.73
LPAcaO	682	62	106	30	1	62	11.00	23.37	212.47	15.54			35.16	154.66	0.38
LPAmpO	109	5	97	2	1	5	21.80	42.14	193.30	88.99			74.25	136.23	0.75
LPAraO	743	76	111	27	1	76	9.78	16.68	170.65	14.94			44.78	162.72	0.18
LPChiO	65	2	51	1	1	2	32.50	26.16	80.50	78.46			26.16	80.50	0.91
LPCopO	147	47	19	18	1	47	3.13	3.89	124.28	12.93			11.17	129.22	0.12
LPComO	10005	27	6529	14	1	27	370.56	1251.91	337.85	65.26			1784.95	249.76	0.75
LPDecO	25	5	15	1	1	5	5.00	5.83	116.62	60.00			0.00	0.00	0.54
LPGasO	135	4	116	2	1	4	33.75	54.99	162.92	85.93			85.56	126.76	0.82
LPHemO	215	30	77	14	1	30	7.17	15.09	210.60	35.81			23.39	152.30	0.66
LPIsoO	36	2	31	1	1	2	18.00	18.39	102.14	86.11			0.00	0.00	0.87
LPOpiO	77	5	63	2	1	5	15.40	26.74	173.61	81.82			34.65	89.99	0.94
LPOrtO	927	3	884	3	1	3	309.00	498.39	161.29	95.36			498.39	161.29	0.97
LSAcaO	4626	28	1924	17	1	28	165.21	426.70	258.27	41.59			701.17	257.67	0.78
LSAraO	1889	88	535	18	1	88	21.47	70.36	327.78	28.32			156.33	148.96	0.48
LSCopO	1340	52	437	23	1	52	25.77	84.12	326.44	32.61			178.37	292.85	0.62
LSComO	3789	7	2152	5	1	7	541.29	804.69	148.66	56.80			1037.01	136.84	0.52
LSHemO	2349	52	597	14	1	52	45.17	120.32	266.36	25.42			318.78	189.99	0.39
LSLepO	250	34	38	18	1	34	7.35	8.55	116.31	15.20			24.99	129.97	0.56
LSOrtO	125	3	122	2	1	3	41.67	69.57	166.98	97.60			69.57	166.97	0.95
LSThyO	1934	9	1431	3	1	9	214.89	477.53	222.22	73.99			1295.42	133.96	0.81
LSTrhO	88	7	52	4	1	7	12.57	18.03	143.39	59.09			27.40	124.54	0.48
MPAraO	1064	90	271	28	1	90	11.82	34.94	295.55	25.47			90.62	238.47	0.21
MPCopO	371	59	84	16	1	59	6.29	12.87	204.67	22.64			29.36	126.62	0.37
MPAcaO	1099	72	114	31	1	72	15.26	26.81	175.64	10.37			58.51	165.04	0.44

Taxa	NoInd	NoSp	NoCom	NoFam	NoOrd	Sp/HigherT	Ind/Sp	SDInd/Sp	CVIndSP	%Com	SDSp/Fam	CVSp/Fam	SDInd/Fam	CVInd/Fam	Rho
MPComO	7567	28	3786	14	1	28	270.25	728.16	269.44	50.03			1009.66	186.80	0.56
MPHemO	410	40	223	18	1	40	10.25	35.17	343.12	54.39			73.13	321.06	0.47
MPTrhO	160	1	160	1	1	1	160.00		0.00	100.00				0.00	1.00
MPAmpO	381	4	303	2	1	4	95.25	142.94	150.07	79.53			267.99	140.68	0.64
MPTrlO	54	7	32	2	1	7	7.71	10.93	141.69	59.26			26.87	99.52	0.56
MPGasO	65	6	28	4	1	6	10.83	10.30	95.08	43.08			16.01	98.52	0.53
MPChiO	213	2	181	2	1	2	106.50	105.36	98.93	84.98			105.36	98.93	0.89
MPDipO	45	5	23	2	1	5	9.00	8.37	93.00	51.11			0.71	3.16	0.58
MPOrtO	2279	5	2264	4	1	5	455.80	1010.82	221.77	99.34			1129.18	198.19	0.99
MPIsoO	34	3	29	2	1	3	11.33	15.37	135.62	85.29			22.63	133.12	0.79
MPOpiO	44	5	25	2	1	5	8.80	10.26	116.59	56.82			12.73	57.86	0.64
MPLepO	27	14	9	10	1	14	1.93	2.16	112.00	33.33			2.91	107.78	0.27
MSAraO	1631	82	289	17	1	82	19.89	51.58	259.32	17.72			134.15	139.83	0.26
MSCopO	402	45	138	23	1	45	8.93	21.64	242.24	34.33			41.81	219.48	0.31
MSAcaO	2509	32	1075	18	1	32	78.41	224.22	285.97	42.85			324.88	233.07	0.76
MSComO	2178	6	1845	5	1	6	363.00	737.17	203.08	84.71			801.57	184.02	0.68
MSHemO	2602	81	857	16	1	81	32.12	126.99	395.32	32.94			279.60	171.93	0.52
MSThyO	1350	10	823	3	1	10	135.00	280.37	207.68	60.96			927.72	137.44	0.78
MSOrtO	100	6	80	4	1	6	16.67	31.08	186.48	80.00			42.03	168.12	0.70
MSLepO	191	22	81	12	1	22	8.68	17.83	205.37	42.41			32.25	202.62	0.50